Evaluating the Influence of Flooding on Aquatic Food-webs in Basins of the Peace-Athabasca Delta Using Isotopic Tracers

by

Stephanie Karrie-Lynn Lyons

A thesis
presented to the University of Waterloo
in fulfilment of the
thesis requirement for the degree of
Master of Science
in
Biology

Waterloo, Ontario, Canada, 2010

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

I understand that my thesis may be made electronically available to the public.

Abstract

Periodic flooding has been widely believed to serve an important role in maintaining water levels and productivity of aquatic basins in floodplain landscapes. Here, I analyze four basins of contrasting flood frequencies (one through-flow, one pulse-flooded, two non-flooded) and two adjacent river sites in the PAD were sampled during the open-water season of 2007 and spring of 2008 to characterize linkages between hydrological processes (using O and H stable isotopes) and limnological conditions, and to assess how these linkages affect trophic interactions involving the aquatic flora and fauna (using C and N stable isotopes). The water balance and water chemistry of the through-flow basin was dominated at all times by the input of river water which reduced concentrations of nutrients and ions. In contrast, evaporation played an important role in the water balance and concentrated nutrients and ions in the non-flooded basins. Surprisingly, pulse-flood events had short-lived effects on the water balance and carbon stable isotopic signatures of biota. Hydrological and limnological conditions in the pulse-flooded basin were similar to those of the river water shortly after spring flooding. After flooding, evaporation caused rapid increase of $\delta^{18}O$ of the water comparable to patterns observed in the non-flooded basins, but recovery of water chemistry variables was delayed. In the non-flooded and pulse-flooded basins, $\delta^{13}C_{DIC}$ declined due to atmospheric CO_2 invasion under conditions of high primary productivity and pH that generated strong kinetic fractionation. This decline in $\delta^{13}C_{DIC}$ values produced the opposite effect compared to when photosynthesis occurs under non-limiting carbon conditions, as occurred in the through-flow basin. This feature provides important new knowledge to improve paleolimnological interpretation of $\delta^{13}C$ values of organic matter in sediment cores to track past changes in flooding regimes. Importantly, this study shows that pulse floods

exert short-lived transient (~1-2 months) effects of the water balance and carbon dynamics of aquatic food-webs and do not elevate aquatic production, but exert longer lasting (at least an entire open-water season) on water chemistry conditions. This contrasts with previous beliefs that the effects of pulse flooding are more profound and longer lasting.

Acknowledgements

Obtaining this degree has been a long anticipated achievement for me. There are many people who I would like to thank for their support and guidance. First, I would like to thank my supervisors Roland Hall and Brent Wolfe for accepting me for the opportunity to conduct research in the PAD. I am very fond of my time that I spent in Fort Chipewyan and the opportunities that I have had during my time at University of Waterloo under your supervision.

Thank you to Robert Grandjambe for sharing several laughs, knowledge and assistance while conducting fieldwork. Our time together provided me with memories that I enjoy sharing with friends and family.

I would like to acknowledge the help of two other Masters students, Caleb Light and Matt Ennis, for assisting me in collecting samples and being willing to explore the PAD with me.

I am extremely thankful to my family and friends who have supported me on this long and winding quest. I know I have missed several celebrations as my education endeavours have taken me on various journeys. The stories I come back with and the experiences have all contributed to where I want to go in life.

Thank you to Environment Canada NLET and UW Environmental Isotope Lab for analyzing my samples. I appreciate your assistance with my research.

I would also like to acknowledge the funding sources for my research, without which I would not have been able to celebrate completing this thesis. The funding sources for this study were provided by grants from the Natural Sciences and Engineering Research Council of Canada (NSERC) Northern Research Chair Program, NSERC Discovery Program and Northern Research Supplement Grant, and the Northern Scientific Training Program of Indian and Northern Affairs Canada. I thank the staff of Wood Buffalo National Park who provided logistical support for fieldwork and flood monitoring maps. Also, I would like to thank the University of Waterloo for the Travel Bursary which provided me the opportunity to attend conferences to share knowledge and network with other researchers.

To everyone who I have met along the path of this journey – thank you.

Table of Contents

List of Figures

List of Tables

Introduction

Ecological conditions within floodplain basins are strongly regulated by hydrologic processes including river flooding. A predictable sequence of physical, chemical and biological changes occurs in floodplain basins that experience regular flooding (Lewis et al. 2000). Initially, flood waters that are turbid with suspended sediments reduce light penetration through the water column, but they also supply sufficient nutrients to stimulate growth of phytoplankton. Once floodwaters recede, mineral turbidity decreases which leads to proliferation of phytoplankton, macrophytes and epiphytes (Tockner et al. 1999; Junk 2005). Thus, floodplains and the associated aquatic ecosystems are often highly sensitive to temporal and spatial variability of hydrological processes (i.e. flood regimes) since floodwaters provide a primary source of water, nutrients and suspended materials (Junk 2005; Roozen 2005; Wantzen 2008).

Growing concerns over impacts of climatic- and human-induced changes on river flow regimes and flooding stimulate the need to improve scientific methods to detect, quantify and predict changes in ecological conditions of floodplains (Squires et al. 2002; Schindler and Smol 2006; Lindholm et al. 2007). There is much known about ecological response to flooding in deltas and floodplains where flooding occurs on a fairly regular basis (e.g., Junk 2005, Roozen 2005, Wantzen et al. 2008), but knowledge is scant for systems where flooding regimes are more variable. The Peace-Athabasca Delta (PAD) is among the floodplains where it is important to understand the effects of changing flooding regimes because recent studies suggest it is likely to experience reduced flooding as a consequence of climate change and upstream utilization of water resources (Wolfe et al. 2008b). Knowledge from other floodplains is not adequate to predict ecological consequences of changes in flood

regimes on basins of the PAD, because the PAD possesses important differences compared to floodplains that have been more extensively studied in South America (e.g., Amazon) and Europe (e.g., Rhine Delta). In these systems, seasonal flooding of rivers elevates the concentration of dissolved nutrients in floodplain basins (Junk 2005; Roozen 2005; Wantzen et al. 2008). The PAD differs from floodplains in South America and Europe because PAD basins span a broad spectrum of flood frequency (continuous to >2 decades between floods) and river water is more dilute in dissolved nutrients (but not particulate nutrients) than the basin.

The PAD is an important feature in the Canadian landscape for historical, cultural and ecological reasons. It is situated in northern Alberta and is part of Canada's largest national park, Wood Buffalo National Park. This 3900 km^2 freshwater inland delta has formed at the western end of Lake Athabasca (59^{o}N 112^{o}W) where the Peace, Athabasca and Birch rivers converge (Figure 1). It has received international recognition as a UNESCO (United Nations Educational, Scientific and Cultural Organization) World Heritage Site and Ramsar wetland (International Ramsar Convention on Wetlands). The numerous shallow basins and river network provide important feeding, staging and breeding habitats for residential and migrating waterfowl, fish and mammals (including the largest free-ranging herd of woodland bison). The delta has cultural significance for First Nations communities who have historically depended on this delta for sustenance through fishing, trapping and hunting. This northern delta is an important ecosystem under threat of multiple stressors, including reduced river drainage due to climate warming, water extraction for oil sands processing, and flow regulation (Schindler and Donahue 2006; Wolfe et al. 2008ab).

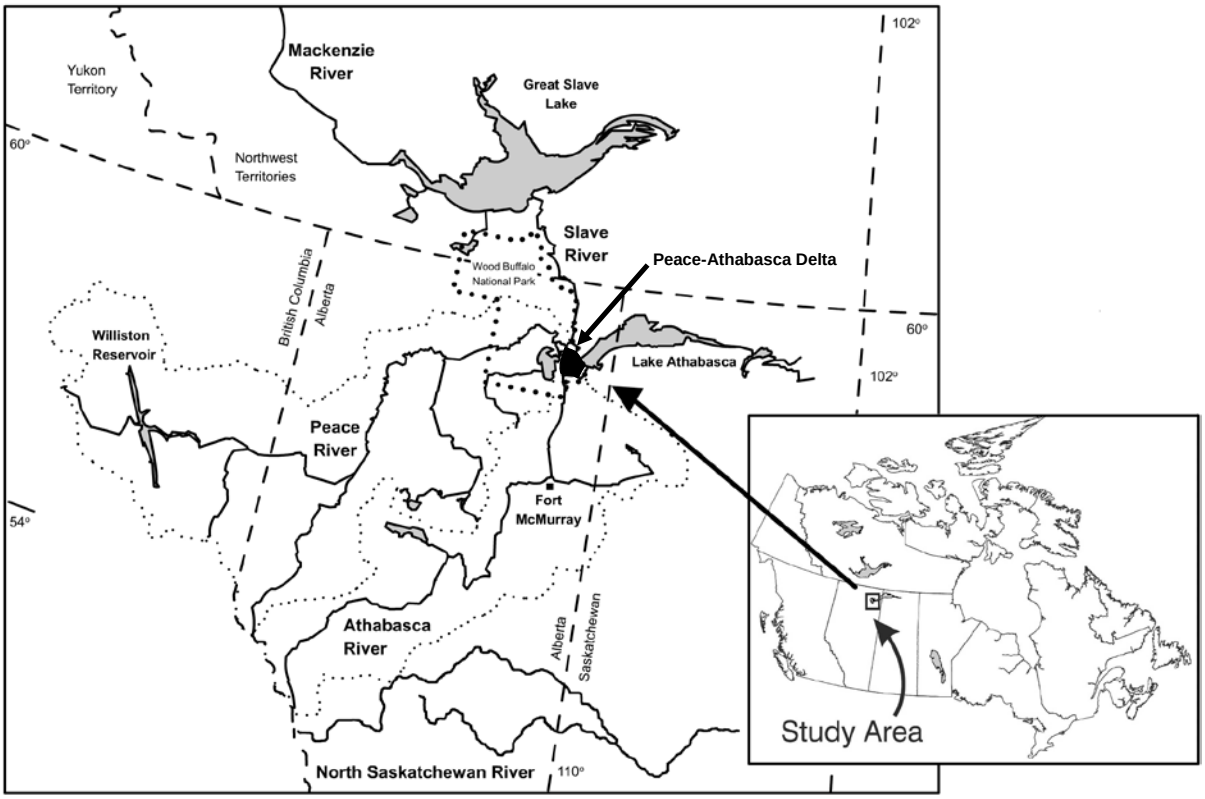


Figure 1. Location of the Peace-Athabasca Delta at the west end of Lake Athabasca in north-eastern Alberta, Canada.

Research conducted in the PAD over the past 40 years has focussed on understanding hydrological conditions and the responses of the delta floodplain to hydrological changes. Initial research was driven by concerns that regulation of the Peace River for hydroelectric production, which began in 1968, would cause declines in flood frequencies and water levels in the delta (PAD-PG 1973). Structural engineered efforts, including rock weir dams, have been implemented to attempt to maintain water levels, flooding regimes and ecological integrity (PAD-IC 1987). It was later recognized that high river flows during the open-water season could not generate widespread flooding, but instead that ice-jam floods and headwater runoff generation during the spring freshet were the main factors promoting extensive delta-wide flooding (Prowse and Conly 1998; Peters et al. 2006). Paleolimnological studies demonstrated that the delta has naturally experienced significant fluctuations in water levels

and flood regimes, in response to climate-driven changes in the quantity and seasonality of river discharge (Wolfe et al. 2008b).

Several reports have concluded that floods are essential to maintain water levels and biological productivity of this delta landscape, including the ponds and lakes, but the conclusions were not all based on scientific data (PADPG 1973; Townsend 1975; Prowse and Demuth 1996; Prowse and Conly 1998). In fact, data remains scant that can assess the roles and importance of changes in flood regime on the aquatic food-webs in floodplain basins of the PAD. However, such information is important to help anticipate future food-webs responses to declines in headwater flow generation and increasing consumptive water use (Schindler and Donahue 2006).

A preliminary study by Köster et al. (2008) suggested that flooding exerts important influence on aquatic food-web structure in aquatic basins of the PAD. Their study compared carbon and nitrogen stable isotope compositions in abiotic and biotic components of the aquatic food-webs of two hydrologically different basins, based on one-time sampling of biota in September 2005. Basin PAD 1 had not received floodwaters for at least two decades, whereas PAD 31 had received pulses of flood water during the spring and summer of at least the past six years. The results of the study indicated the carbon and nitrogen isotope compositions of various food-web components differed between the flooded and non-flooded basins. In addition, there was temporal variability in the isotope composition of dissolved inorganic carbon in PAD 1 between July and September. Köster et al. (2008) suggested the need to increase the number of basins to span a broader range of flooding regimes, as well as more frequent sampling, to further explore the influence of flooding on aquatic food-webs. Their study also suggested the inclusion of analysis of benthic

macroinvertebrates since the benthic food-web can provide important nutrient cycling roles in shallow basins (e.g., Solomon et al. 2008).

Objectives

This research builds upon the preliminary work of Köster et al. (2008) by evaluating the influence of flooding on aquatic food-webs using a broader gradient of river-influenced basins within the PAD. Limnological variables and stable isotope tracers of hydrological (using hydrogen and oxygen isotopes) and food-web components (using carbon and nitrogen isotopes) were analyzed to determine the role of flooding on the aquatic food-webs of four lentic systems within the PAD. Specifically, the study design compares two basins in the PAD that received floodwaters (PAD 31, 45) with two basins that did not flood in spring 2007 and 2008 (PAD 1, 3) to determine how flooding influences limnological conditions and food-webs. The non-flooded basins, to our knowledge, have not flooded since at least 1997. The two flooded basins differed in their flood regimes. PAD 45 (Mamawi Lake) receives continuous river inflow from Athabasca River via Mamawi Creek, whereas PAD 31 (Johnny Cabin Pond) receives periodic pulses of floodwater during the spring and summer months.

This study is intended to contribute new knowledge concerning the ecological conditions within the PAD. Understanding of contemporary carbon and nitrogen isotope composition of abiotic and biotic components of PAD basins will help determine the role and importance of flooding as a factor in regulating ecological conditions. Additionally, the knowledge gained about the response of carbon and nitrogen isotope composition of aquatic biota to flooding will assist with the interpretation of past hydroecological changes from paleolimnological studies. Overall, the study contributes useful information for the effective stewardship of this important delta in the face of climatic variability and river flow alteration through the understanding of how aquatic food-web components within the PAD basins will likely respond to changes in flood frequency.

Site Descriptions

Seasonal and interannual patterns of variation in physical, chemical and biological conditions were assessed at each basin from samples collected during the last two weeks during months of May, July and August of 2007 and May of 2008. Selection of the four basins was based on several years of research and observations during previous hydroecological studies (Wolfe et al. 2007), as well as on consideration of sufficient access by boat and hiking to permit the sampling.

'PAD 1' (local name: 'Devil's Gate Pond'; 58º48.4'N, 111º14.7'W) and 'PAD 3' (unofficial name; 58º49.9'N, 111º17.2'W) are two small shallow (~1.1 m and 0.6 m water depth, respectively) closed-drainage basins that have not, to our knowledge, flooded since at least 1997 (Figures 2, 3, 4). They do not receive or exchange water with adjacent basins or the river. Both basins are located in the Peace sector of the delta adjacent to the Rivière des Rochers. Dense macrophyte cover, dominated by *Potomogeton*, *Myriophyllum* and *Utricularia* species, develops between June and September. The perimeters are surrounded by sedges and cattail. Dense willow shrubs grow adjacent to the ponds, and spruce and poplar tree stands dominate the higher relict levees along the river and rock outcrops.

'PAD 31' (local name: 'Johnny Cabin Pond'; 58º29.8'N, 111º31.2'W) is a small shallow (~1.4 m maximum water depth) flood-prone basin located in the Athabasca sector of the delta (Figures 2, 3). PAD 31 flooded due to ice-jams in spring of 2007 and 2008, and during a period of open-water high river flow in the early summer of 2007 (Figures 4, 5). There is no active water exchange with adjacent open-water bodies except during spring and summer high-water events along Mamawi Creek. This basin has a floating perimeter of sedges and cattail surrounded by dense willow shrubs in the low-lying regions. Submerged

vegetation is dominated by *Potomogeton* and *Myriophyllum* species. The vegetation on higher terrain (i.e., river levees) is dominated by spruce and poplar trees. Paleolimnological records analyzed from this pond indicate that flood frequency increased markedly after 1982 when a natural geomorphological event (Embarras River Breakthrough) diverted a portion of the Embarras River flow into Mamawi Creek that flows along the margin of PAD 31 (Wolfe et al. 2008a). Mamawi Creek was an inactive channel prior to the Embarras Breakthrough, but now carries ~6 % of the Athabasca River flow and the proportion is expected to increase over time (PADTS 1996).

'PAD 45' (local name: Mamawi Lake; 58º31.7N, 111º24.5W; Figures 2, 3) is a large open-drainage lake that continuously receives river water mainly from Mamawi Creek, and discharges mainly through the Chenal des Quatre Fourches. Also, during seiche events on Lake Athabasca, lake and river water can back-flow into PAD 45 via Chenal des Quatre Fourches. The basin has a large surface area yet it is shallow (<2 m maximum water depth) and remains highly turbid during most of the open-water season. Submerged macrophytes were observed to proliferate in late summer and consisted predominantly of *Potomogeton* and *Myriophyllum* species. This is the only one of the study basins where fish are present.

The two river sites (Mamawi Creek and Rivière des Rochers, sample identification: 'PAD RMAM' and 'R11', respectively; Figure 2, 3) were sampled to assess the influence of floodwaters on basin water chemistry. Mamawi Creek is a distributary of the Embarras River, flows adjacent to PAD 31 and discharges into Mamawi Lake. Rivière des Rochers carries Lake Athabasca outflow water, passes by PAD 1 and PAD 3, and discharges into the Slave River.

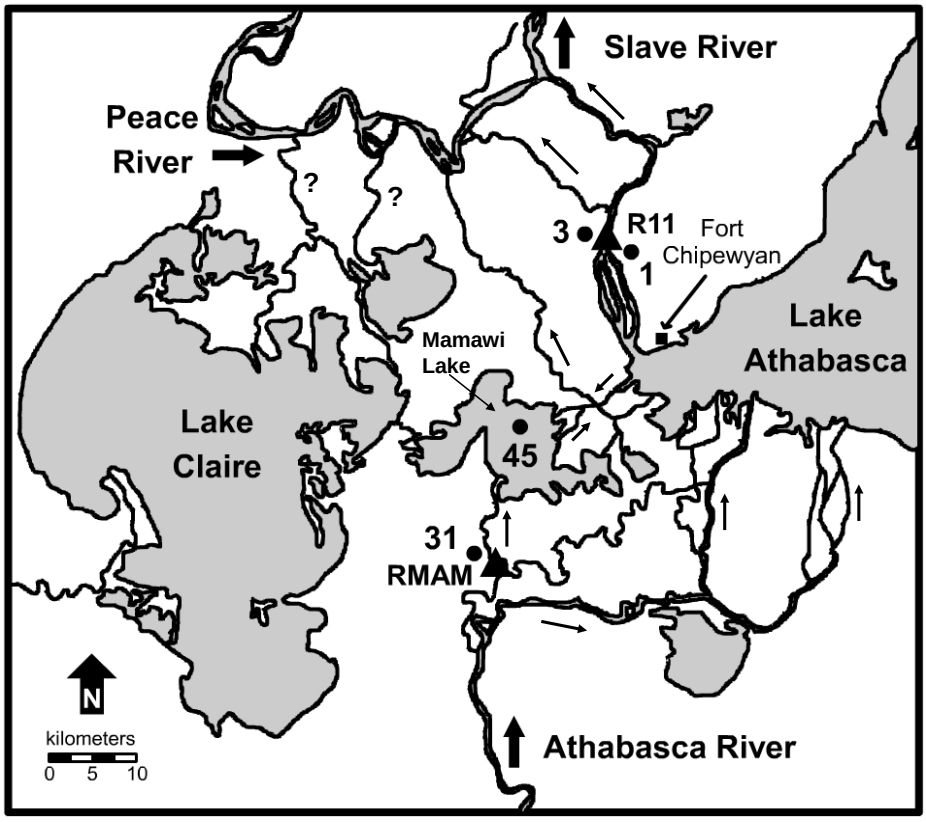
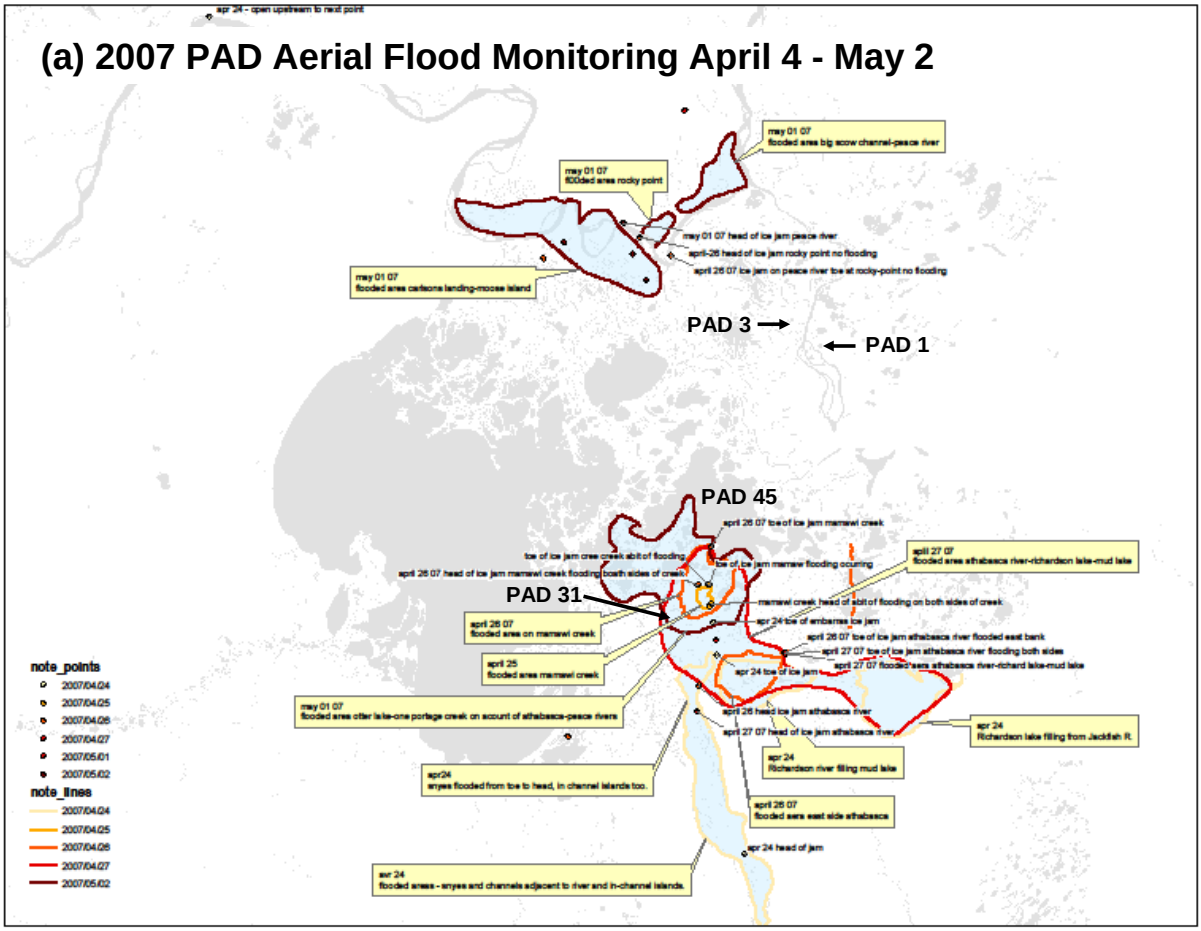


Figure 2. Map of the Peace-Athabasca Delta showing locations of the study sites. The numbered closed circles represent the basins that were sampled and closed triangles indicate the river sites (each indicated with an identification code beginning with the letter 'R'). Drainage routes through the PAD are indicated by arrows.

Figure 3. Aerial views of the four study basins and two adjacent rivers sampled during the 2007-08 open-water seasons. The sampled basins include varying frequency of river connection – non-flooded (PAD 1 and 3), pulse-flooded (PAD 31 – flooded in spring and early summer of 2007 and spring of 2008) and through-flowing (PAD 45 – constantly receives river discharge).


(a) 2007 PAD Aerial Flood Monitoring April 4 - May 2
apr 24 - open upstream to next point
may 01 07
flooded area big acow channel-peace river
may 01 07
flooded area rocky point
may 01 07 head of ice jam peace river
april-26 head of ice jam rocky point no flooding
april 26 07 ice jam on peace river toe at rocky-point no flooding
may 01 07
flooded area carlsons landing-moose island
PAD 3
PAD 1
PAD 45
april 26 07 toe of ice jam marnawi creek
toe of ice jam cree creek abit of flooding
toe of ice jam marnaw flooding occuring
april 26 07 head of ice jam marnawi creek flooding both sides of creek
april 27 07
flooded area athabasca river-richardson lake-mud lake
PAD 31
marnawi creek head of abit of flooding on both sides of creek
april 26 07
flooded area on marnawi creek
apr 24 toe of embarras ice jam
april 26 07 toe of ice jam athabasca river flooded east bank
april 27 07 toe of ice jam athabasca river flooding both sides
april 25
flooded area marnawi creek
apr 24 toe of ice jam
april 27 07 flooded aera athabasca river-richard lake-mud lake
may 01 07
flooded area otter lake-one portage creek on account of athabasca-peace rivers
april 26 head ice jam athabasca river
april 27 07 head of ice jam athabasca river
apr 24
Richardson lake filling from Jackfish R.
apr 24
Richardson river filling mud lake
apr24
snyes flooded from toe to head, in channel islands too
april 26 07
flooded aera east side athabasca
apr 24 head of jam
apr 24
flooded areas - snyes and channels adjacent to river and in-channel islands.
note_points
2007/04/24
2007/04/25
2007/04/26
2007/04/27
2007/05/01
2007/05/02
note_lines
2007/04/24
2007/04/25
2007/04/26
2007/04/27
2007/05/02

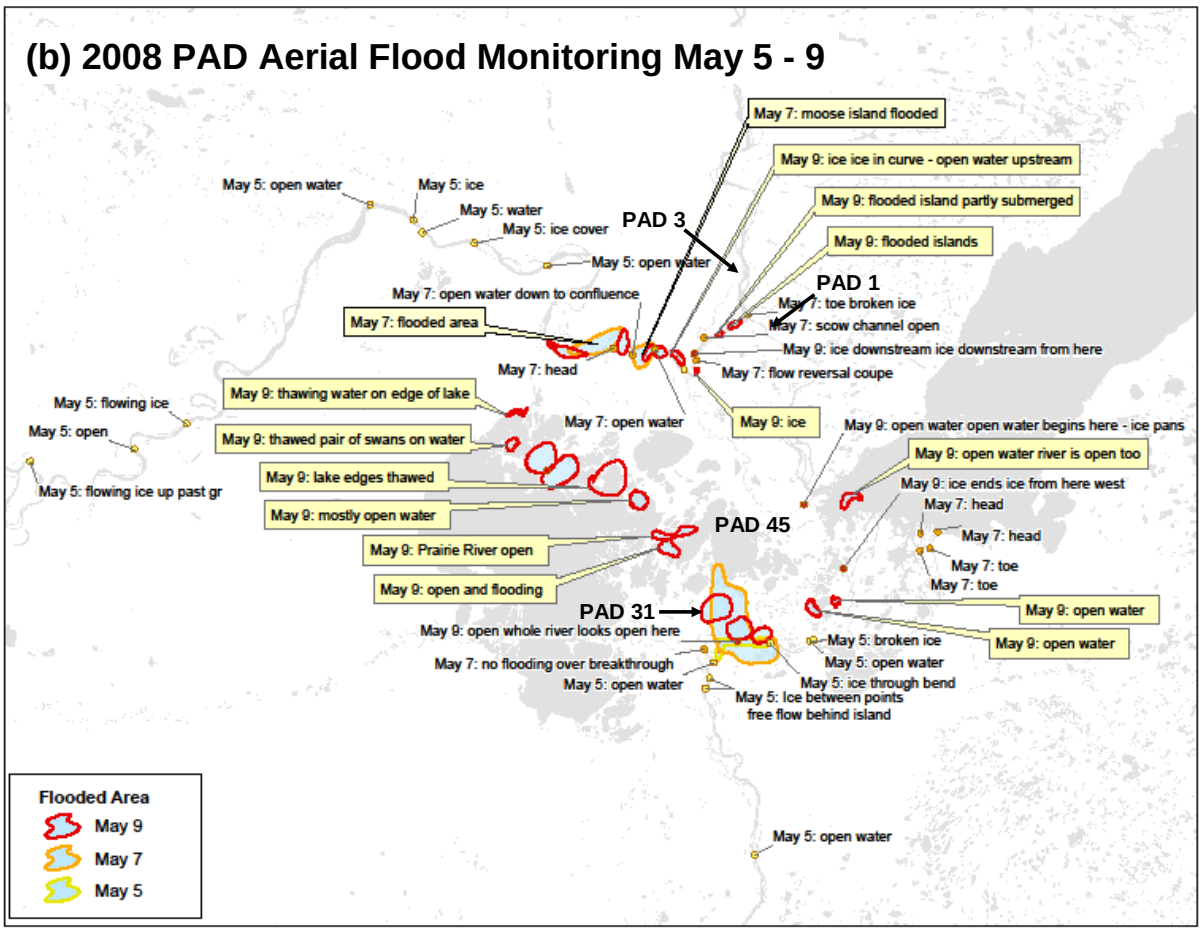


Figure 4. Spring flood monitoring maps for (a) 2007 and (b) 2008 for Wood Buffalo National Park and the Peace-Athabasca Delta indicating PAD 31 was flooded in the spring of both years (Wood Buffalo National Park 2007 and 2008).

Figure 5. Observed flooding in PAD 31 during May 2007 and 2008 sampling. Notice the standing water on top of terrestrial catchment land (in the foreground) that provides evidence of recent floodwaters.

Field Methods

Hydrology and Water Chemistry

During each of the four sampling campaigns, several water column variables were measured *in situ* near the central region of each basin to assess spatial and temporal relationships among the hydrological and water chemistry characteristics. Measurements of pH were obtained *in situ* at 10-cm intervals through the water column using a handheld YSI 660 MDS multiparameter meter. Three light level (μmol photons $m^{-2}s^{-1}$) measurements were obtained at each 10-cm interval through the water column using an Apogee QMSS-S model light meter. The mean of the three values at each depth was natural log-transformed and plotted versus depth (m) to determine the extinction of photosynthetically active radiation (PAR) through the water column (k_dPAR). Additionally, changes in water depth were continuously recorded every half hour between May and late August of 2007 using Tru Track Ltd. WT-HR water-level data loggers.

Samples of water and biota were gathered at each study location during the four sampling periods. Water samples were obtained from approximately 30 cm below the water surface near the central region of each basin and river site. Samples for hydrogen and oxygen isotope analyses were collected by completely filling 30 mL bottles and capping bottles tightly to prevent evaporation. Water for chemical analyses was pre-screened using a 63-μm mesh to remove large zooplankton and other particles. Averaged concentrations of total suspended solids (TSS) were estimated for each sample site by filtering two known volumes of sample water through pre-ashed, pre-weighed Whatman GF/C (1.2 μm pore size) glass fibre filters upon return to the research base. Chlorophyll *a* concentrations were estimated for each site using GF/F glass fibre filters with 0.7 μm pore size to filter known

volumes of sample water at the research base in Fort Chipewyan. The filters with samples for TSS and chlorophyll *a* analyses were frozen until analysis.

Food-webs

Water samples were obtained at each site to characterize sources of carbon and nitrogen (it was later discovered that insufficient volumes were obtained to permit analysis of dissolved inorganic nitrogen) to primary producers. Samples for analysis of $\delta^{13}C$ of dissolved inorganic carbon (DIC) were collected *in situ* using a 125 mL glass serum bottle with rubber stopper. Samples for dissolved organic carbon (DOC) and dissolved inorganic nitrogen (DIN) were obtained by pre-filtering sample water through a 63-µm mesh, passing the water through a pre-combusted quartz filter with pore size 0.45 µm, and keeping the filtrate. All the air in the DIC, DOC, and DIN bottles was extracted using a syringe needle, and the bottles were sealed air-tight with electrical tape and stored cool (4ºC) until analysis.

Samples of phytoplankton, zooplankton and epiphytes for food-web analyses were harvested at each sample location using similar methods to those of Köster et al. (2008). Although Köster et al. (2008) used vertical net hauls to collect the biota, we used horizontal tows to collect phytoplankton and zooplankton. Vertical hauls are typically used in deeper water bodies to compensate for daily migration of various zooplankton species however, the basins in this study are shallow (< 2 m). Phytoplankton and zooplankton were sampled using multiple tows with nets having 25-µm and 63-µm mesh sizes, respectively. The phytoplankton samples were filtered through the 63-µm zooplankton net to remove zooplankton and provide a distinct size class (i.e., 25 µm < x < 63 µm), as suggested by Köster et al. (2008). Macrophytes were harvested from a canoe using a rake to obtain

macrophyte material growing above the sediment surface. Collected macrophytes were brushed lightly with a toothbrush and washed with deionized water to remove epiphytes. The remaining epiphyte suspension was filtered through a 48-µm mesh screen to remove any zooplankton. Benthic invertebrates were obtained using an Ekman grab sampler and sieved through a 63-µm mesh. Snails were collected by hand when observed in the field. Fish were not collected for isotopic analyses because we did not have approval to collect fish for research purposes.

Different preservation techniques were used for the invertebrates, filters and plant material before analysis for stable isotope composition. To allow gut clearance, zooplankton samples were stored for two hours in water from the basin or river that had been filtered through cellulose acetate filters with 0.45-µm pore size. Benthic invertebrate samples were frozen until the sample could be processed to separate organisms from the remaining organic matter. The zooplankton and benthic invertebrates (excluding snails) were then preserved in 60% ethanol (final concentration) to prevent sample degradation while waiting to be processed for analysis (Sarakinos et al. 2002; Feuchtmayr and Grey 2003). Shells were removed from the snails and the tissue was air-dried at 60ºC in an oven. Phytoplankton and epiphyte samples were collected on pre-ashed Whatman quartz filters with 0.45-µm pore size. The quartz filters were folded in half, wrapped in aluminum foil and frozen until analysis.

Laboratory Methods

Hydrology and Water Chemistry

Analysis of alkalinity (ALK), conductivity and concentrations of major ions (Cl^-, SO_4^{2-}, Ca^{2+}, Mg^{2+}, Na^+, K^+), DOC, SiO_2, and nutrients (total nitrogen (TN), NO_2^{-2} + NO_3^-, NH_3^+, total phosphorus (TP)) were performed by Environment Canada's National Laboratory for Environmental Testing using methods from Environment Canada (1996). All samples for chemical analysis were kept at ~4ºC until analyzed. Concentrations of suspended solids (total, inorganic and organic suspended solids indicated as TSS, ISS and OSS, respectively) were determined by using standard techniques (APHA 2000). Chlorophyll *a* measurements were analysed at University of Waterloo using the Furgal and Smith (1997) pigment extraction flourometric method. Duplicate samples were analyzed and results averaged for determination of suspended solid and chlorophyll *a* contents. Data from the water level loggers were plotted and standardized to depths taken by a calibrated plumb bob.

Isotope Hydrology

Analyses of oxygen and hydrogen isotope composition in water samples were conducted by the Environmental Isotope Laboratory at the University of Waterloo (UW EIL) using standard methods on the IsoPrime Micromass Mass Spectrometer (Drimmie and Heemskerk, 1993; Morrison et al., 2001). Results are expressed as delta (δ) values, which represent the per mil (‰) deviation from a specified standard. Calculation of the isotopic abundance within a sample is as follows:

$$\delta \text{ value } (‰) = [(R_{sample}/R_{standard}) - 1] * 1000$$

where R represents the $^{2}H/^{1}H$ or $^{18}O/^{16}O$ ratios in both the sample and standard. Results are reported relative to the international standard Vienna Standard Mean Ocean Water (V-SMOW) for both hydrogen and oxygen stable isotopes. The $\delta^{18}O$ and $\delta^{2}H$ results were normalized for Standard Light Antarctic Precipitation to -55.5 ‰ and -428 ‰, respectively (Coplen 1996). The analytical uncertainties are ±0.2 ‰ for $\delta^{18}O$ and ±2.0 ‰ for $\delta^{2}H$. For the purpose of this thesis, only the $\delta^{18}O$ values are reported in conjunction with water level data. The $\delta^{2}H$ values are in the Appendix.

Food-webs

In the present study, the invertebrates were generally separated to family level and assigned to trophic groups (according to Merrit and Cummings 1996) as outlined in Table 1. Further separation to genus or species level was implemented when there was an organism known to have a different eating strategy than the majority of family members (according to Merrit and Cummins, 1996; Table 1). A dissecting microscope was used to sort specimens of various family or genus groups until sufficient abundance was obtained for isotopic analysis (~0.25-0.3 µg). Sample material was dried in vials at 60ºC and ground to powder before being weighed into tin capsules for isotope analysis.

Table 1. Zooplankton and benthic invertebrates included within the food-web trophic components.

Primary producers	Primary Consumers	Primary Predators	Secondary Predators	Omnivores
Epiphyte algae Phytoplankton	**Epiphyte/detritus grazers** Ephemeroptera Gastropoda **Detritus grazers** Gammarus Hyalella **Benthic grazers** Oligochaeta **Pelagic grazers** Bosmina Calanoida Cladocera Cyclopoida	Chaoborus Hydracarina Polyphemus pediculus	Hemiptera Hirudinea Odonata	Chironomidae Trichoptera

Köster et al. (2008) speculated that the presence of carbonate enriched the carbon isotope values of epiphyte samples in their food-web study. To assess the influence of carbonates on isotopic composition of the phytoplankton and epiphyton samples, results were compared from a subset of seven filtered samples that were acidified with values prior to being acidified. A portion of each filter was required for non-acidified isotopic analysis. The remaining section of each filter was gassed with concentrated HCl in an air-tight vessel for at least 5 hours and oven-dried following methods of del Giorgio and France (1996), Pinnegar and Polunin (1999), and Sierszen et al. (2006). Carbonate contamination did not appear to be a factor in changing carbon isotope values (Appendix 13) and therefore non-acidified values were used in food-web analysis.

Carbon and nitrogen isotope analyses of food-web samples were conducted at UW EIL using Continuous Flow Isotope Ratio Mass Spectrometry (CF-IRMS) [Thermo-Finnigan Delta Plus coupled to a Carlo Erba 1108 Elemental Analyzer]. Results are reported as $\delta^{13}C$ and $\delta^{15}N$ values in per mil (‰), with respect to the international standards Vienna Peedee

Belemnite marine limestone (V-PDB) for carbon and atmospheric nitrogen (AIR) for nitrogen. The analytical uncertainties are ± 0.2 ‰ for $\delta^{13}C$ and $\delta^{15}N$.

Numerical Analyses

Variation in limnological and hydrological variables between the flooded and non-flooded ponds was assessed using Principal Components Analysis (PCA). All limnological variables except for pH and k_dPAR were transformed using ln(x+1) to improve normality and equalize variances. This transformation was used by Wiklund et al. (2010) for a larger data set within the same study area. Data for NO_3^-, NO_2^- and NH_3^- were omitted from numerical analysis because many values were below detection limits. Limnological parameters for PAD 1 in May 2008 were not used for numerical analysis because some of the data were not reliable. River samples were added passively to the PCA to assess the influence of river waters on flooded basins without affecting the relationships of limnological conditions among the study basins. The data were then scaled for inter-variable distances and variables were divided by the standard deviation and centred prior to ordination. Ordination by PCA was performed using CANOCO version 4 (ter Braak and Šmilauer, 1998).

Results

Hydrology

Integration of information on seasonal patterns of water-level fluctuations and $\delta^{18}O$ of basin water identifies that hydrological processes differed among the non-flooded basins (PAD 1, 3), pulse-flooded basin (PAD 31) and through-flow basin (PAD 45) during 2007 (Figure 6). In all basins, water levels were typically highest in the early spring following snowmelt, and generally declined over the course of the summer. The magnitude of water-level drawdown in the non-flooded basins was smaller compared to that of the pulse-flooded basin and corresponded with evaporative increases in $\delta^{18}O$ (Figure 6). In the two non-flooded basins, $\delta^{18}O$ values reflect the absence of river connection as the basin $\delta^{18}O$ values remain separated from the river $\delta^{18}O$ values. Maximum ^{18}O-enrichment for PAD 1 and 3 occurred in early August, with values surpassing the climate normal flux-weighted estimate of the terminal basin steady-state isotope value (δ_{SSL}), as derived in Wolfe et al. (2007).

Two high-water events (late May, mid June) along Mamawi Creek resulted in flooding of PAD 31, which raised the basin water levels (Figure 6). The input of isotopically-depleted river floodwater into PAD 31 was reflected by the low $\delta^{18}O$ composition of PAD 31 in May. The May $\delta^{18}O$ value is similar to that of the adjacent river. A second flood event occurred in PAD 31 during 11-12 June 2007, which elevated water levels in PAD 31 by 38 cm and delayed evaporative ^{18}O-enrichment of the basin water compared to the non-flooded basins. During the May 2007 flood event, PAD 31 had lower $\delta^{18}O$ composition than the non-flooded basins by approximately 6 ‰. After these flood pulses, PAD 31 became disconnected from the adjacent river and the pattern of water-level decline due to evaporation was similar to that of the two non-flooded basins. The $\delta^{18}O$

values also diverged away from the river $\delta^{18}O$ values by up to ~4 ‰ as the open-water season progressed, but they remained below the climate normal flux-weighted estimate of the terminal basin steady-state isotope value (δ_{SSL}).

At the through-flow basin (PAD 45), high spring water levels and low $\delta^{18}O$ values were sustained longer during the 2007 open-water season compared to the other basins. The strong and persistent role of river waters on the water balance of PAD 45 is indicated by prolonged maintenance of high water levels and low basin water $\delta^{18}O$ values similar to Mamawi Creek.

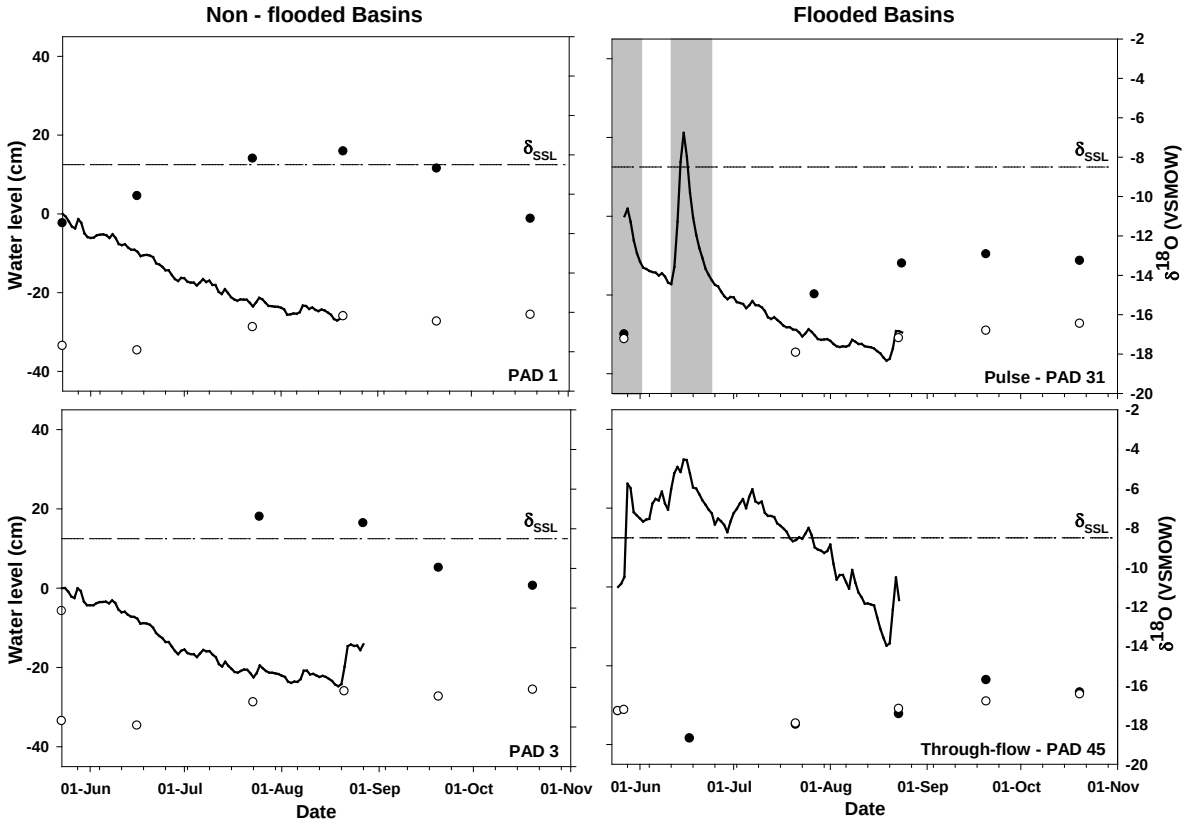


Figure 6. $\delta^{18}O$ values of the basin water (solid circles) and the adjacent river (empty circles) as well as water-level changes (solid line) of the four basins for the 2007open-water season. Shaded bars in PAD 31 indicate periods of flooding. Missing isotope data for PAD 3, PAD 31 and PAD R11 is because samples could not be collected due to logistical reasons.

Limnology

Principal Components Analysis (PCA) of the water chemistry data from the four study basins was used to explore the role of basin hydrology on limnological conditions (Figure 7). Eigenvalues for the first and second axis were 0.662 and 0.157, respectively, which explained 81.9 % of the variation in water chemistry within the dataset. The first PCA axis separated the basins into the three hydrological types (through-flow, pulse-flooded, and non-flooded). The non-flooded ponds (PAD 1, PAD 3), positioned at the left end of axis 1, were positively correlated with concentrations of most ions, TKN, dissolved P, DOC, and pH, alkalinity and conductivity. Sample scores for the through-flow basin (PAD 45) were positioned at the right end of axis 1 near the river samples, associated with high concentrations of suspended solids, SO_4, TP and dissolved Si, and low water transparency (= high k_dPAR). Sample scores collected from the pulse-flooded basin (PAD 31) in spring after receiving flood waters were positioned to the right along PCA axis 1 near those of the rivers and through-flow basin, but subsequently shifted to the left along PCA axis 1 to a position intermediate between those of the through-flow and non-flooded basins. The second PCA axis generally captured seasonal variation of water chemistry conditions within each basin (Figure 7). Sample scores from the early season (May) at each basin plotted lowest along PCA axis 2 and generally increased with time during the 2007 season.

Water chemistry conditions in the pulse-flooded basin (PAD 31) experienced the greatest fluctuations over the 2007 season, as indicated by the spread of sample scores across both axes of the PCA ordination. They overlapped with sample scores of the through-flow basin (PAD 45) and the rivers in May of 2007 and 2008, shortly after receiving flood inputs. Once PAD 31 was disconnected from the adjacent river, however, the sample scores

diverged along axes 1 and 2 to a unique location in the PCA ordination space as the water chemistry started to shift towards conditions similar to those of the non-flooded basins. The shift in water chemistry resulted in PAD 31 plotting high on axis 2 and intermediate along axis 1 between the non-flooded basins and the through-flow basin and rivers by July. This was due to an unusual combination of high pH and concentrations of dissolved P and low alkalinity and concentrations of suspended solids, DIC, DOC, TN, chlorophyll *a* and most ions (except high SO_4). The pH, and concentrations of dissolved P, and suspended solids in PAD 31 during July and August were similar to the conditions in the non-flooded basins. However, alkalinity and concentrations of DIC, DOC, TN, chlorophyll a, and ions (excluding SO_4) in PAD 31 during July and August were similar to the conditions in the trough-flow basin.

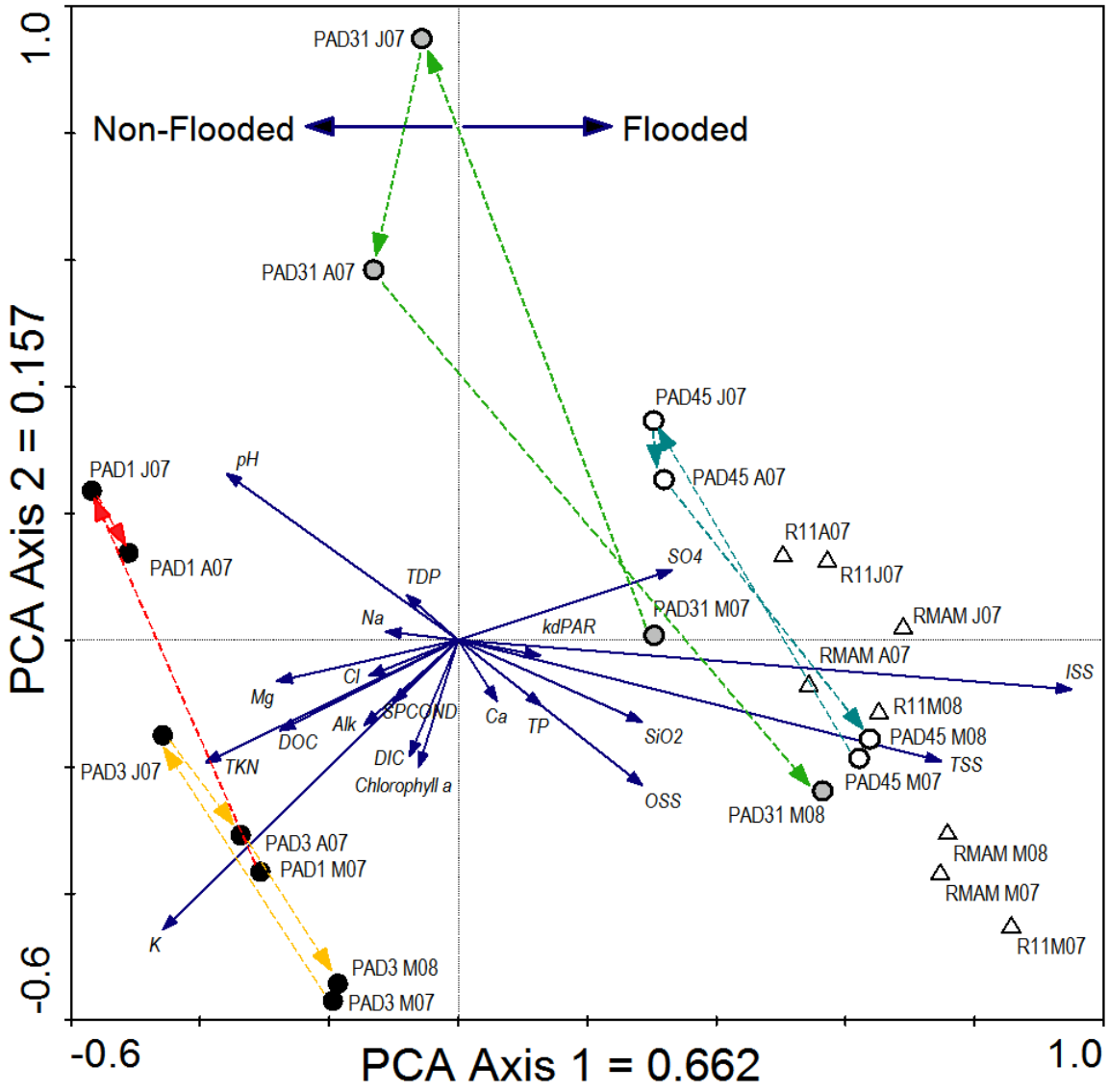


Figure 7. Principal components analysis (PCA) ordination diagram showing differences in water chemistry conditions between the flooded and non-flooded study basins of the Peace-Athabasca Delta (n=4). The eigenvalues are indicated along both axes. Samples from the rivers (open triangles) were plotted passively. Sample scores of the non-flooded basins (PAD 1, PAD 3) are illustrated with solid circles, whereas open circles indicate the through-flow basin (PAD 45) and grey-filled circles indicate the pulse-flooded basin (PAD 31). The hatched lines indicate the direction of seasonal change for each basin. Codes for the environmental vectors are listed in Table 2. The letter and number codes beside each sample score identify the month (M = May, J = July, A = August) and year (07 = 2007, 08 = 2008) the samples were collected. May 2007 results for PAD 1 were not included in analysis.

Table 2. Identification of the codes used to identify the limnological variables used in the PCA ordination diagram (Figure 7).

Variable code	Variable
Cl	Chloride
SO4	Sulfate
Ca	Calcium
Mg	Magnesium
Na	Sodium
K	Potassium
DOC	Dissolved Organic Carbon
DIC	Dissolved Inorganic Carbon
SiO2	Silica dioxide
Alk	Alkalinity
SPCOND	Specific Conductivity
TKN	Total Kjeldahl Nitrogen
TP	Total Phosphorus
TDP	Total Dissolved Phosphorus
TSS	Total Suspended Solids
ISS	Inorganic Suspended Solids
OSS	Organic Suspended Solids
kdPAR	Extinction coefficient of photosynthetically active radiation through the water column

The non-flooded basins had high concentrations of most ions, nutrients, DOC, pH and conductivity, high water clarity (=low k_dPAR) and low concentrations of suspended solid concentrations relative to flooded basins and river water (Figure 8). Concentrations of magnesium and potassium were two to five times higher in the non-flooded than the flooded basins throughout the sample period (Figure 8g and i). River water contribution to PAD 45 and PAD 31 diluted the concentration of TKN by approximately three times compared to the non-flooded basins (Figure 8m). Dissolved organic carbon concentrations in the non-flooded basins were up to three times greater than in the flooded basins throughout the sample period (Figure 8l). In the non-flooded and pulse-flooded ponds, pH rose by at least two pH units between May and July. In contrast, pH remained relatively constant in the through-flow

basin and rivers (Figure 8a). Conductivity levels were also different between the hydrological types (i.e., non-flooded ponds ranged between 297 to 387 µS/cm and flooded ponds ranged between 200 to 256 µS/cm; Figure 8d). The suspended solid concentrations in the two non-flooded basins were consistently below 10 mg/L with a temporary increase of organic solids in the spring (i.e., 4.35 mg/L OSS of the 5.45 mg/L TSS; Figures 8s and u). Water clarity was relatively high in both non-flooded basins and there were low concentrations of suspended solids (Figures 8r and s). There was an increase in summer k_dPAR values likely due to shading from aquatic macrophytes.

In contrast to the non-flooded basins, the through-flow basin (PAD 45) had water chemistry similar to the river water (Figure 8). PAD 45 was characterized by relatively high concentrations of SO_4, TP, suspended solids, and dissolved SiO_2, high k_dPAR, and low concentrations of most nutrients and ions, and low conductivity. Concentrations of SO_4 in PAD 45 ranged from 5 mg/L higher in the spring to 15 mg/L higher in the summer compared to the non-flooded ponds (Figure 8k). Total phosphorus concentrations in PAD 45 were almost double in May compared to the other basins, and subsequently declined to values similar to the other three basins during the remainder of the 2007 season as river flows declined (Figure 8p). In May, the dissolved P concentration was similar to the other basins and then became lower in PAD 45 compared to the non-flooded basins for the remainder of the sample period (Figure 8q). Elevated concentrations of dissolved Si in PAD 45 were maintained throughout the season due to continuous supply of suspended solids from the river (Figure 8j). Suspended solid concentrations were highest in PAD 45 with the majority consisting of inorganic solids (i.e., 76 mg/L ISS of the 85 mg/L TSS; Figures 8s and t). High

k_dPAR values (low water transparency) occurred in the water column due to the high concentration of suspended solids (Figures 8r and s).

Limnological conditions varied more over the four sample periods in pulse-flooded PAD 31 than in all other basins. The most pronounced limnological effects from flooding occurred in the spring when the water chemistry was most similar to that of the through-flow basin and rivers. Spring flooding resulted in diluted concentrations of most ions (Ca, Cl, Mg, and K), DIC, and nutrients, and a short-lived increase in suspended solids (Figure 8). This basin had similar limnological conditions to the adjacent river and PAD 45 during flooding in both 2007 and 2008. Once the basin was hydrologically disconnected from the adjacent river, some limnological variables diverged from patterns of PAD 45 and started to converge on values of the non-flooded basins (i.e., pH; Figure 8a), yet most values of variables remained distinct from those of the non-flooded basins (e.g, lower alkalinity and concentrations of TN, DOC, DIC, Ca, Mg, and SiO_2; higher concentrations of SO_4; Figure 8b, c, e, g and j). Concentration of TDP remained elevated throughout the growing season compared to the through-flow basin (Figure 8q). Suspended solids rapidly settled out of the water column after flooding as indicated by the drop of concentrations to values similar to the non-flooded basins (Figure 8s).

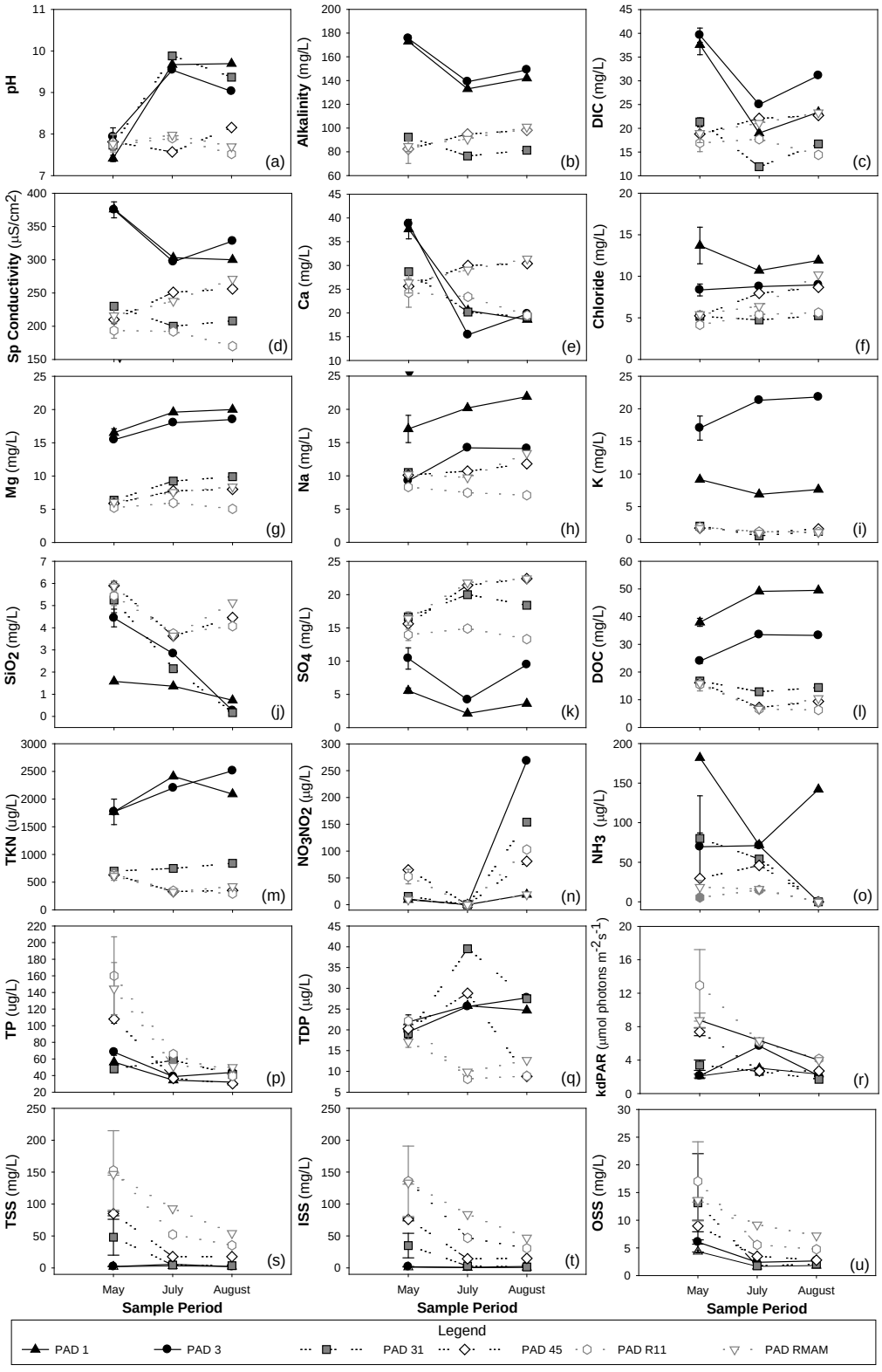


Figure 8. Temporal changes in water-quality variables of the study basins and rivers. The value presented for May is the average of the 2007 and 2008 data. The vertical line for May values represent the maximum and minimum values collected during in 2007 and 2008.

Effects of flooding on carbon cycling, pH and phytoplankton

Limnological properties associated with the different hydrological basin types are reflected in the associated DIC and chlorophyll *a* concentrations as well as the pH (Figure 9a to c). In May, when the through-flow and pulse-flooded basins received river water, the concentration of DIC was half of that in the non-flooded basins. The through-flow basin maintained relatively consistent DIC concentrations throughout the open-water season. By July, the DIC concentrations in the non-flooded and pulse-flooded basins declined to half of the values in May. Maximum phytoplankton biomass (represented by chlorophyll *a*) for the non-flooded basins occurred in spring, yet the pulse-flooded and through-flow basins had relatively low phytoplankton biomass throughout the season. By August, all the basins had similar standing crop of phytoplankton (chlorophyll *a* ~5 µg/L). In the spring, all the basins had pH values between 7 and 8. During the summer, the pH rose to values above 9 in the non-flooded and pulse-flooded basins, whereas pH remained lower and relatively consistent pH (~7.5-8) throughout the open-water season in the through-flow basin.

Carbon isotope composition of DIC and phytoplankton were likewise distinct among the hydrological basin types (Figure 9d, e). In May, the non-flooded basins had high $\delta^{13}C_{DIC}$ values (~ -5 ‰) compared to the flooded ponds (~ -10 ‰). From May to July, the $\delta^{13}C_{DIC}$ in the non-flooded basins was reduced by 7 ‰ and then increased by ~2 ‰ in August. The pulse-flooded basin displayed a similar temporal pattern in the $\delta^{13}C_{DIC}$ to that of the non-flooded basins but values were consistently lower by 5 ‰ due to the initial supply of lighter $\delta^{13}C_{DIC}$ from river water in the spring. Phytoplankton carbon isotope composition had different seasonal patterns than the $\delta^{13}C_{DIC}$. All basins had low $\delta^{13}C_{phytoplankton}$ values in May, ranging between -34 ‰ to -26 ‰. By July, the $\delta^{13}C_{phytoplankton}$ values increased by 13 and 8

‰ in the non-flooded and pulse-flooded basins, respectively. In contrast, the $\delta^{13}C_{phytoplankton}$ values in the through-flow basin remained relatively constant throughout the open-water season (-28.3 to -26.7 ‰).

The difference between $\delta^{13}C_{DIC}$ and $\delta^{13}C_{phytoplankton}$ (i.e., $\Delta^{13}C_{DIC\text{-}phytoplankton}$) provides an estimate of the carbon isotope fractionation (Fry 2006), and it strongly varied among the hydrological basin types (Figure 9f). Fractionation values in May were higher in the non-flooded basins (24 to 29.3 ‰) than the flooded basins (16 to 19 ‰). By July, fractionation for the non-flooded and pulse-flooded basins declined by at least 15 ‰. However, the through-flow basin increased slightly over the season by 5 ‰.

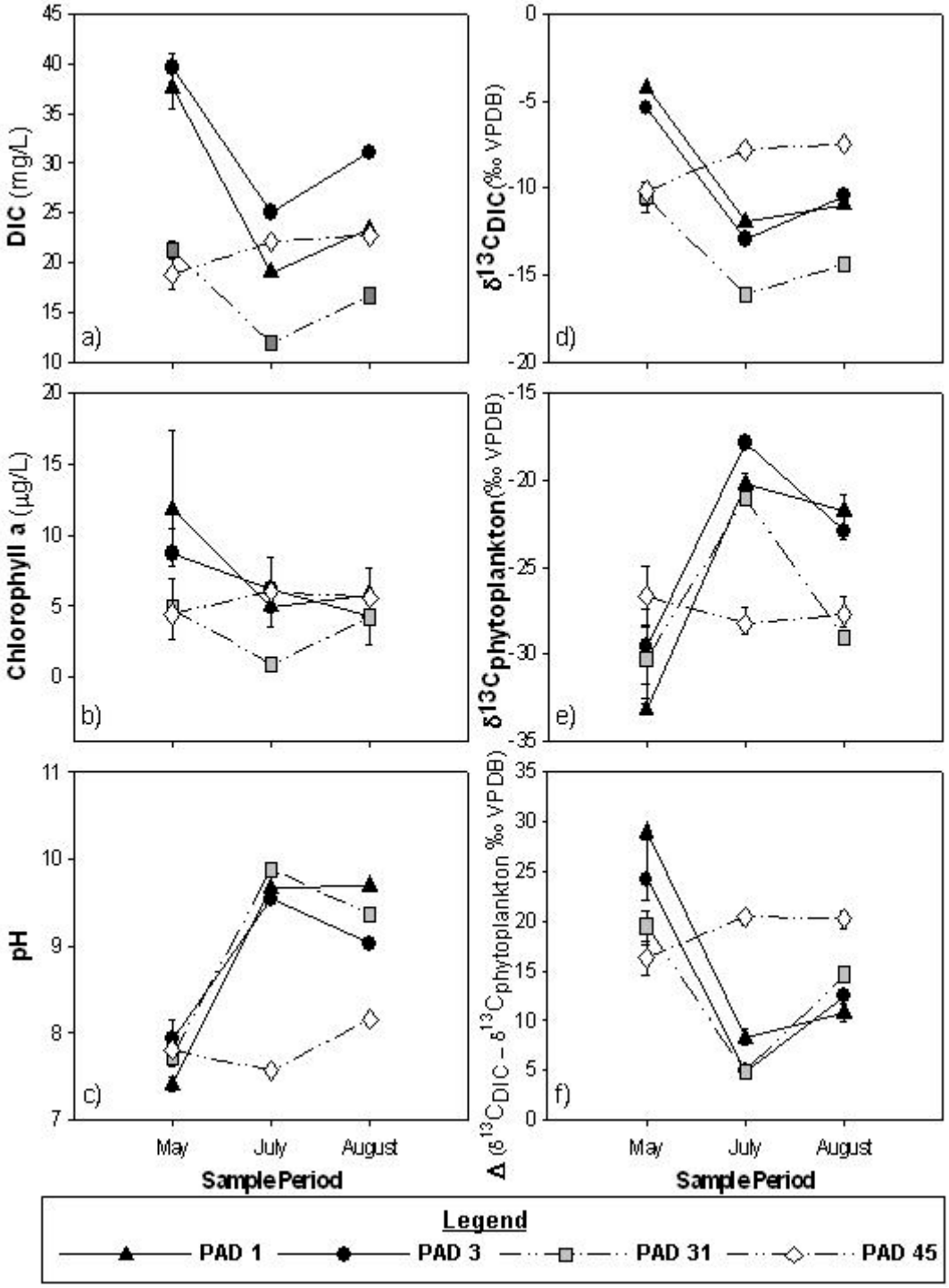


Figure 9. Seasonal changes in the study basins of concentrations of DIC and chlorophyll *a*, and pH, $\delta^{13}C$ of DIC and phytoplankton, and DIC-phytoplankton carbon isotope fractionation. Values for May are presented as the mean of 2007 and 2008, with vertical bars indicating the maximum and minimum values in 2007 and 2008.

Carbon and nitrogen isotope compositions of aquatic food-webs

Biota for the analysis of aquatic food-webs in the four basins included pelagic, epiphytic and benthic components. Based on the samples collected, the top trophic level consisted of insects. These basins are generally too shallow to maintain sufficient dissolved oxygen levels during ice-covered periods to support fish. The through-flow basin is the only site of the four that maintains a fish population throughout the year due to the constant supply of water from the rivers and the ability of fish to migrate to Lake Athabasca and river channels. Some young of the year fish were observed in the pulse-flooded basin in May 2007 but they would not likely survive the winter.

Carbon isotope composition of the various food-web components displayed seasonal shifts, some of which were distinct among the hydrological basin types (Figure 10). The $\delta^{13}C$ values of the various trophic levels typically increased from May to July. The non-flooded and pulse-flooded basins experienced the greatest degree of ^{13}C-enrichment in the detritus grazer, phytoplankton, pelagic grazer, and omnivore groups. The $\delta^{13}C$ values increased by 4.7 to 13.1 ‰ for these trophic groups between May and July. There was a general tendency for the $\delta^{13}C$ values in these same components to drop by 1 to 8 ‰ from July to August. The through-flow basin typically showed far less ^{13}C-enrichment from May to July and for several trophic groups (detritus grazers, phytoplankton, pelagic grazer, secondary predator) the $\delta^{13}C$ values became more depleted. In contrast to the non- and pulse-flooded basins, the $\delta^{13}C$ values of most of the trophic groups in the through-flow basin increased slightly (~1 to 3 ‰) from July to August.

Although seasonal patterns of carbon isotope composition were distinct among the hydrological basin types for organisms near the base of the pelagic food-webs

(phytoplankton, pelagic grazers), they were less distinct for the uppermost trophic level (secondary predator) and for biota that feed from other resources (e.g., benthos, epiphytes, detritus) (Figure 10). The lowest pelagic trophic levels (phytoplankton and pelagic grazers) showed the greatest difference in $\delta^{13}C$ patterns among the basin types. The $\delta^{13}C$ values of pelagic grazers closely followed that of the phytoplankton. The $\delta^{13}C$ values of the detritus grazers show similar patterns as the phytoplankton and pelagic grazers but with less ^{13}C-enrichment from May to July. In contrast, the $\delta^{13}C$ values of benthic grazer, epiphytes, epiphyte and detritus grazer, omnivore, primary predator and secondary predator trophic groups did not display patterns that were distinct among the hydrological basins types.

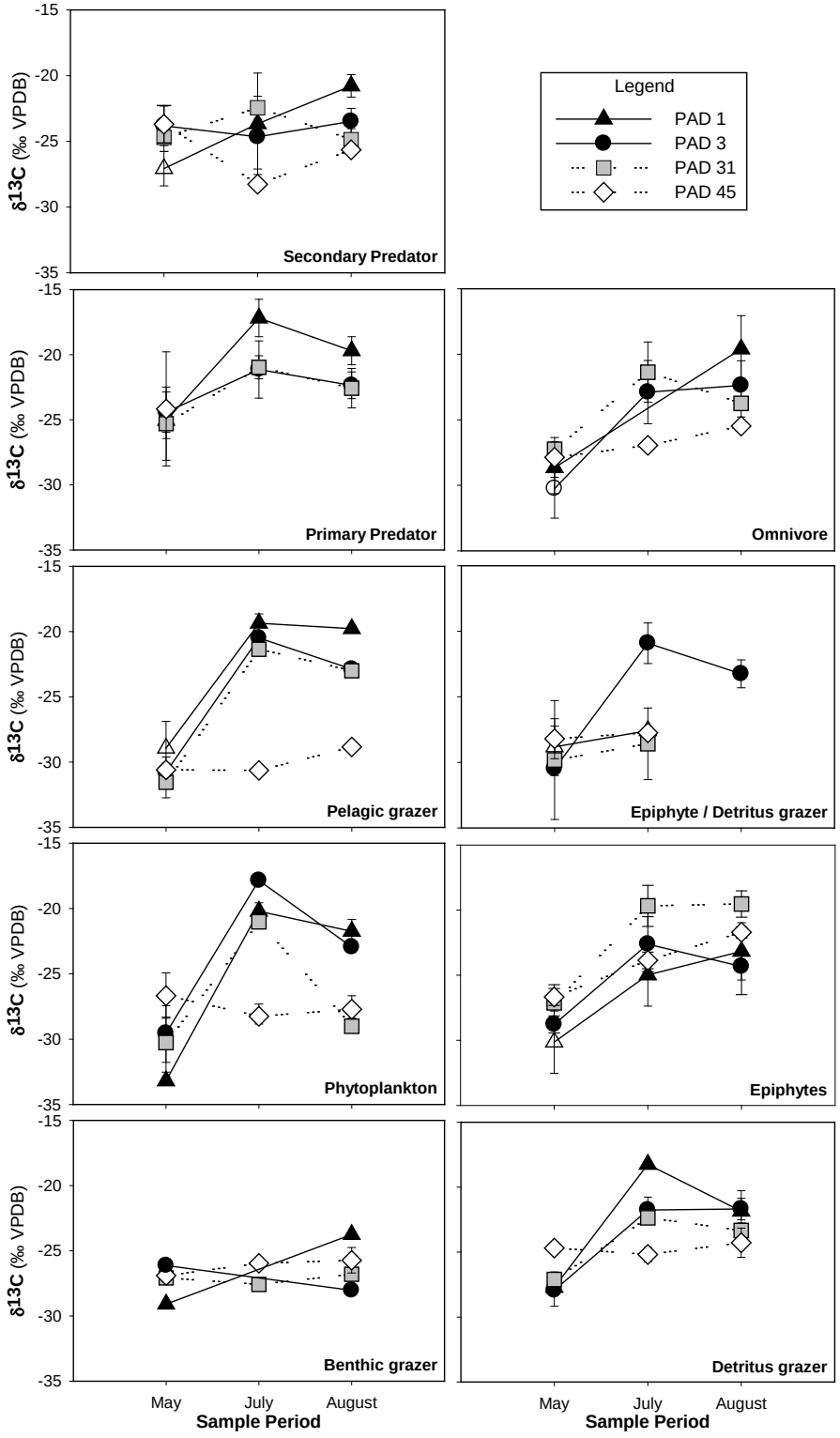


Figure 10. Comparison of seasonal variations in the $\delta^{13}C$ composition of distinct trophic levels in the food-webs of the four hydrologically different study basins of the Peace-Athabasca Delta. The data are the average value of several different taxa in each trophic category (see Table 1 for details) and the vertical bars are the standard errors.

In contrast to the seasonal trends in $\delta^{13}C$, $\delta^{15}N$ values generally declined, particularly between May and July (Figure 11). Seasonal changes in $\delta^{15}N$ values for several of the trophic levels (including detritus, epiphyte, pelagic grazer, omnivore and primary predator) had the most marked declines (≤ 4 ‰) between May and July. Between July and August, the pelagic grazer, primary predator and secondary predator trophic groups tended to increase in $\delta^{15}N$ by 1 to 3 ‰.

Most of the trophic groups did not demonstrate different temporal $\delta^{15}N$ patterns among the hydrological basin types, although the $\delta^{15}N$ values were generally enriched in most trophic levels of the through-flow basin compared with the non-flooded and pulse-flooded basins, except for phytoplankton, primary predator, and secondary predator trophic groups (Figure 11). Pelagic grazers and omnivores were the only trophic groups to have a distinct separation in $\delta^{15}N$ seasonal patterns among the basin types. Throughout the season, the flow-though basin had slightly greater $\delta^{15}N$ values for the pelagic grazers than the other basins. However, $\delta^{15}N$ values in the non-flooded and pulse-flooded basins differed by 0.2 ‰ in August. Omnivores were the only trophic group to have different seasonal patterns of change between the through-flow basin and the other basins, where $\delta^{15}N$ values increased over time in the through-flow basin and declined in the non-flooded and pulse-flooded basins.

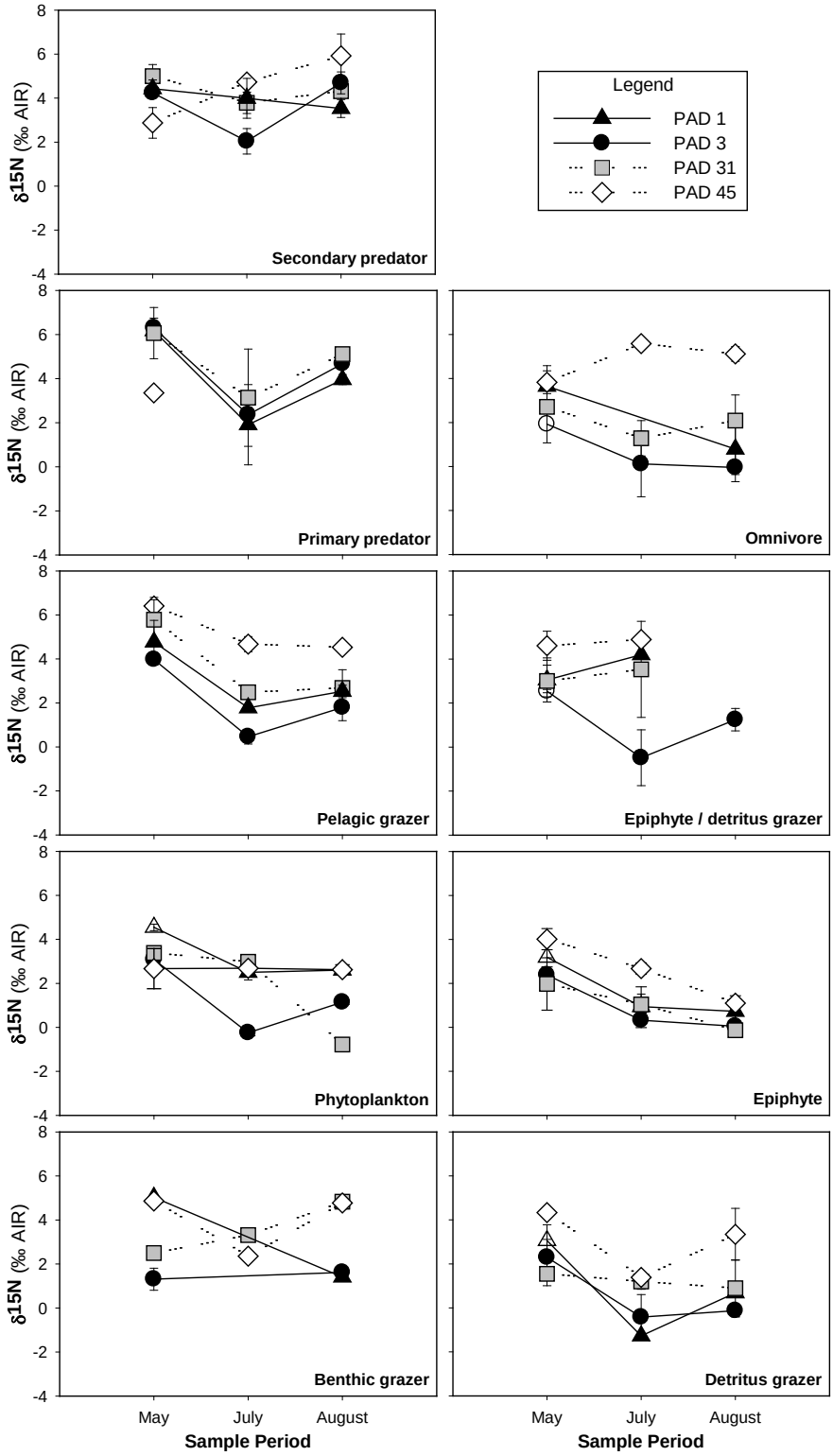


Figure 11. Temporal comparison of $\delta^{15}N$ composition of distinct trophic levels in the food-webs of the four hydrologically different study basins of the Peace-Athabasca Delta. The data are the average value of multiple taxa in each trophic category (see Table 1 for details) and the vertical bars represent the standard error.

Arrangement of trophic groups in conventional $\delta^{13}C$-$\delta^{15}N$ space for the different hydrological basin types revealed seasonal isotopic shifts in the aquatic food-webs (Figure 12). The three hydrological basin types had unique isotopic shifts within the organization of the associated aquatic food-webs throughout the open-water season. In May 2007 and 2008, the aquatic biota in the non-flooded basins typically have the greatest scatter in $\delta^{13}C$-$\delta^{15}N$ space (-34 to -21 ‰ for $\delta^{13}C$ and -7 to -0 ‰ for $\delta^{15}N$, respectively) (Figure 12, rows a and d) when compared to July and August (-23.8 to -19.7 ‰ for $\delta^{13}C$ and -0.1 to 4.7 ‰ for $\delta^{15}N$, respectively) (Figure 12, rows b and c). The aquatic organisms started to have isotopic separation according to trophic levels within the pelagic, epiphytic, and benthic food-chains in the non-flooded basins as the open-water season progressed. Specifically, biota with low $\delta^{15}N$ values tended to include lower trophic levels (epiphytic algae and benthic invertebrates) and pelagic organisms had slightly higher $\delta^{13}C$ values than those associated with epiphyte and benthic consumption. Omnivores, however, had $\delta^{15}N$ values similar to the epiphytic and benthic organisms. Both predaceous trophic groups (primary and secondary) had the highest $\delta^{15}N$ values (3.5 to 4.7 ‰) by late summer.

The food-web of the pulse-flooded basin did not have isotopic separation of biota that corresponded with their position in the food-web by August as is evident in the non-flooded basins. Phytoplankton had an isotopic composition separate from the rest of the organisms as they had the lowest $\delta^{13}C$ and $\delta^{15}N$ values in the pulse-flooded basin. Also, organisms that typically consume benthic matter had $\delta^{15}N$ values similar to the top trophic level in the pulse-flooded basin, which would not normally occur if benthic consumers are consumed and $\delta^{15}N$ is enriched with each trophic transfer.

In the through-flow basin, trophic level separation amongst the aquatic biota occurred in August. Primary producers (epiphytes and phytoplankton) had the lowest $\delta^{15}N$ values (1.1 to 2.6 ‰), primary consumers (pelagic grazer, epiphyte/detritus grazer, detritus grazer, benthic consumer, and omnivore) had mid-range $\delta^{15}N$ values (3.3 to 5.1 ‰), and the secondary predators had the highest $\delta^{15}N$ value (5.9 ‰) in August.

Shifts in the $\delta^{13}C$ range of biota in the non-flooded and pulse-flooded basin occurred as a response to seasonal $\delta^{13}C_{DIC}$ dynamics in the basins. In spring, there was separation between the average biota $\delta^{13}C$ and $\delta^{13}C_{DIC}$ of ~23 ‰ in the non-flooded basins and ~16 ‰ in the pulse-flooded basin. In July and August, this separation of $\delta^{13}C$ values of the organisms and DIC became reduced to ~10 ‰ on average for the non-flooded basins and ~8 ‰ for the pulse-flooded basin. In contrast, the difference in $\delta^{13}C$ between the aquatic biota and DIC for the through-flow basin remained relatively constant (~17 ‰) during the open-water season.

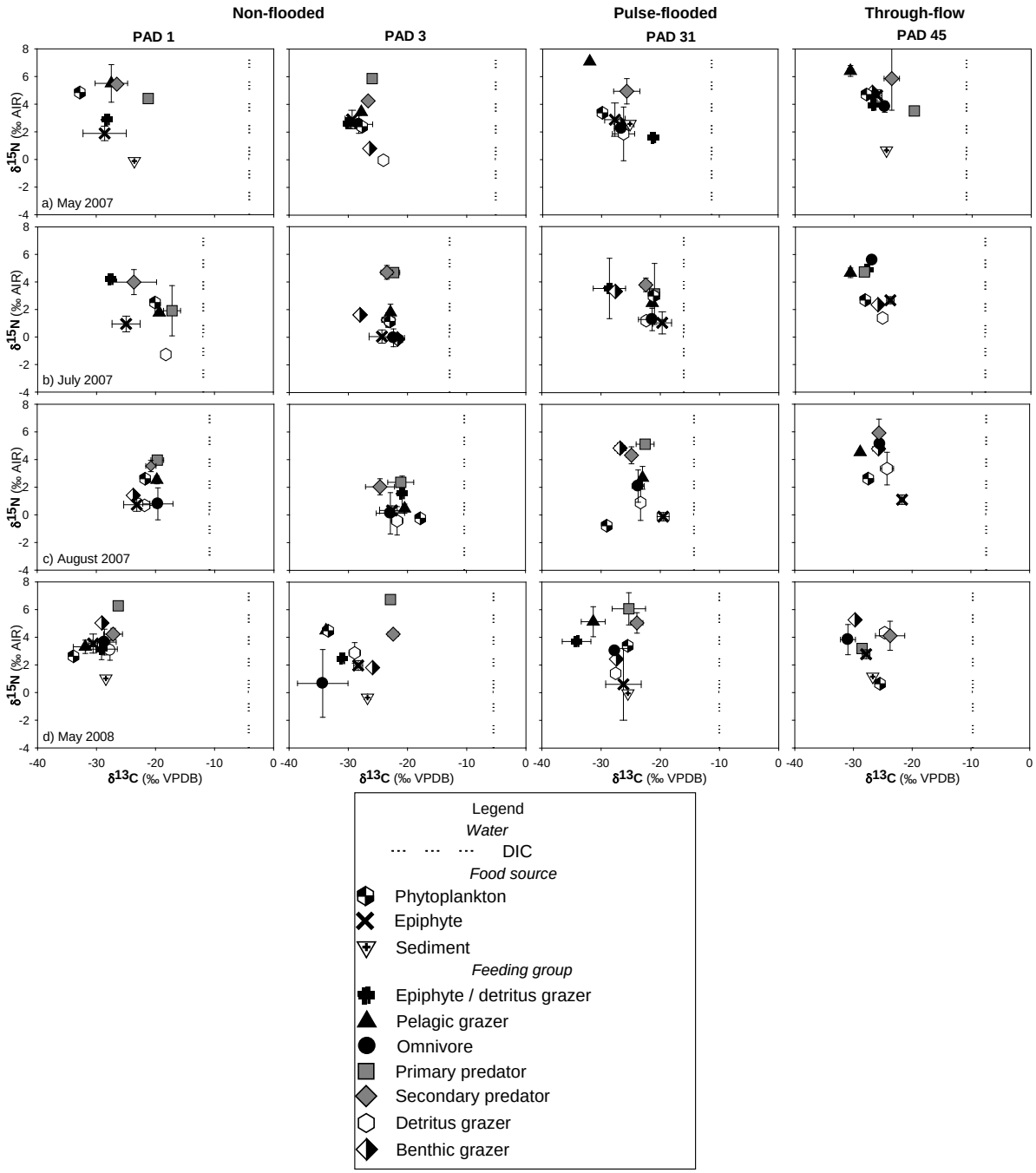


Figure 12. Isotopic variability within the trophic levels of the four basins during the four sample periods during 2007 and 2008. Each value represents an average value of organisms within the feeding category in the associated food-web and the error bars is the standard error. Each row represents the same sample period.

Discussion

Floodwaters are known to exert strong influence on the hydrology as well as physical, chemical and biological conditions of the PAD floodplain basins (Hall et al. 2004; Wolfe et al. 2007; Wiklund et al. 2010). Non-flooded basins had higher spring nutrient and ion concentrations than the rivers and flooded basins (Figure 8) as a result of multiple years of evaporation dominating the water balance of these basins. The role of evaporation in the non-flooded basins is supported by the distinctly higher $\delta^{18}O$ values compared to water of the other basins and rivers throughout the 2007 open-water season (Figure 6). Biological uptake of ions and nutrients may account for decreases in some of the ion and nutrient concentrations during the open-water season, as has been observed in non-flooded basins of the Mackenzie Delta by Lesack et al. (1998). However, concentrations of some nutrients and ions increased typically between July and August, likely due to evaporative concentration and regeneration via decomposition.

In contrast to non-flooded basins, river inputs into the through-flow basin dominated the water balance in 2007, which generated greater water-level fluctuations and similar $\delta^{18}O$ values to the rivers throughout the entire season (Figures 6, 7). Limnological conditions remained similar to those of the rivers throughout the entire season due to the strong influence of inflowing rivers that supplied with water that is dilute in ions and dissolved nutrients, but high in SiO_2 and SO_4 (Figure 7, 8). Rivers supplied a high load of suspended sediments to the through-flow basin, which reduced light penetration into the water column throughout the open-water season compared to the other basins (Figures 7, 8).

Spring and early summer flooding exerted strong but short-lived influence on the water balance of the pulse-flooded basin, however, pulse floods had a longer lasting

influence on water chemistry which endured the entire open-water season. By July, the water balance in the pulse-flooded basin was strongly influenced by evaporation and was comparable to the water balance of the non-flooded basins. After flooding, the water chemistry of the pulse-flooded basin generally remained more dilute (excluding TP, TDP, SiO_2, SO_4, and suspended solids) but followed similar seasonal pattern to those in the non-flooded basins (Figures 7, 8).

Although river flooding plays an important role in regulating hydrological and limnological conditions, the effects on aquatic food-webs of the PAD have remained poorly understood. As explained below, this study shows that differences in basin hydrology exert strong influence on aquatic food-webs, with strongest effects observed at the base of the food-webs. I will first explain the effects of flooding on the carbon isotope patterns with respect to both DIC and phytoplankton. I will then compare the patterns in the carbon and nitrogen isotope composition of the aquatic food-webs in each hydrological basin type.

Effects of flooding on carbon isotope patterns in DIC and phytoplankton

The carbon isotope composition of aquatic plants is determined primarily by the $\delta^{13}C$ of dissolved inorganic carbon (DIC), which is controlled by a dynamic balance of processes. These include isotopic exchange with atmospheric CO_2, input of DIC from the catchment, ^{13}C-enrichment derived from preferential uptake of ^{12}C by primary producers during photosynthesis, and recycling of ^{13}C-depleted carbon from the decay of organic matter in the water column and bottom sediments (Figure 13; O'Leary 1981; Keeley and Sandquist 1992). Typical seasonal behaviour results in summer increase in epilimnion $\delta^{13}C_{DIC}$ due to photosynthesis, whereas deeper waters often contain lower values of $\delta^{13}C_{DIC}$ resulting from

net respiration (e.g., Quay et al. 1986; Wachniew and Rozanski 1997). However, invasion of atmospheric CO_2 into the basin under conditions of high productivity and pH can generate strong kinetic fractionation and a decrease in epilimnion $\delta^{13}C_{DIC}$ values (Herczeg and Fairbanks 1987). Carbon isotope fractionation between dissolved CO_2 ($CO_{2(aq)}$) and the organic substrate may vary strongly, depending on the concentration of $CO_{2(aq)}$ and temperature, and can range from ~ -20 ‰ when $CO_{2(aq)}$ is not limiting to as low as 0 ‰ under $CO_{2(aq)}$-limiting conditions (Deuser et al. 1968; Calder and Parker 1973; Rau 1978; Herczeg and Fairbanks 1987). At low concentrations of DIC and high pH, phytoplankton may incorporate HCO_3^-, which is ~ 8 ‰ higher (at 20°C) in $\delta^{13}C$ than $CO_{2(aq)}$ (Deuser et al. 1968; Emrich et al. 1970; Allen and Spence 1981). Often the balance between photosynthesis and respiration is considered to provide the dominant signal preserved in the $\delta^{13}C$ of organic matter in lake sediment cores with positive excursions frequently interpreted as increased aquatic productivity (McKenzie 1985; Schelske and Hodell 1991, 1995; Dean and Stuiver 1993; Duthie et al. 1996). However, hydrologic change can cause deviations from these established patterns (e.g. Wolfe et al. 1996).

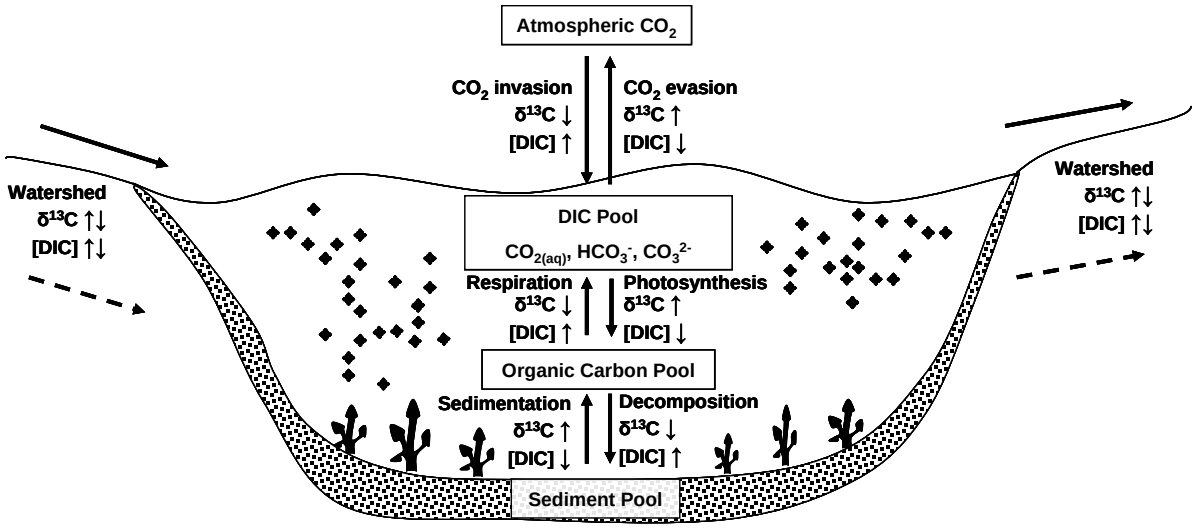


Figure 13. Tracing the potential influences on the carbon cycle with respect to changes in the DIC concentration and isotopic composition in aquatic systems.

The seasonal patterns of $\delta^{13}C_{DIC}$ are distinctly different among the non-, pulse-, and through-flow basins. In the non-flooded basins, $\delta^{13}C_{DIC}$ values were highest in May and declined in July by ~10 ‰ and then increased slightly in August. Higher $\delta^{13}C_{DIC}$ values in the non-flooded basins in May reflect high rates of phytoplankton production (Figure 9b) under conditions in which carbon is not limiting, consistent with high DIC concentrations (Figure 9a). By July, carbon uptake by photosynthesis caused a marked decline in DIC concentration and elevated pH. Macrophytes, rather than phytoplankton account for these changes since phytoplankton represents a very small proportion of the primary producer biomass (~0.33 to 0.41 g/m^3 phytoplankton versus 502 g/m^3 macrophytes based on converting phytoplankton chlorophyll *a* concentrations (Figure 9b) using the equation of Rand et al. 1976 (ug/L = mg/m^3 * 67 * 1g/1000mg = g/m^3) and macrophyte biomass data provided by Wiklund, unpublished data). Although primary production increased between May and July, the $\delta^{13}C_{DIC}$ declined, which is contrary to what is typically observed (e.g., Quay et al. 1986; Wachniew and Rozanski 1997). Removal of isotopically lighter DIC by

the primary producers and invasion of atmospheric CO_2 into the basin would normally be expected to drive the $\delta^{13}C_{DIC}$ to higher values. The decline in $\delta^{13}C_{DIC}$ could not be caused by net respiration of particulate organic carbon because the DIC concentration declined and the pH increased. Likewise, isotopically-depleted DIC from catchment runoff is unlikely as the water levels were decreasing due to net evaporation (Figure 6). Alternatively, a mechanism that could account for the decline in $\delta^{13}C$ is strong kinetic fractionation that can occur during atmospheric CO_2 invasion under conditions of high pH and high primary production. This process, called "chemically enhanced CO_2 invasion", was first described by Herczeg and Fairbanks (1987) in their study of Mohonk Lake, in New York State and has been identified in subsequent studies (Takahashi et al. 1990). A reduction in the intensity of this process may account for the slight rise in $\delta^{13}C_{DIC}$ in August (Figure 9d).

The influence of flooding caused marked deviations in patterns of $\delta^{13}C_{DIC}$ in the flooded basins compared to the non-flooded basins. In both the through-flow and pulse-flooded basins, $\delta^{13}C_{DIC}$ was lower in May, which corresponds with low production. In the through-flow basin, $\delta^{13}C_{DIC}$ increased slightly in July as a consequence of increasing rates of primary production. A continuous supply of DIC from the river prevents carbon limitation and the corresponding isotopic effects of chemically enhanced CO_2 invasion that occurred in the non-flooded basins. This is consistent with far less demand for DIC as evidenced by substantially lower primary production in this basin (phytoplankton: 0.41 g/m^3; macrophytes: 119 g/m^3 using the equation of Rand et al. 1976 and macrophyte biomass data provided by Wiklund, unpublished data) compared to the non-flooded basins. The pulse-flooded basin shows a distinctly different pattern of $\delta^{13}C_{DIC}$ containing features of both the through-flow and non-flooded basins. Initially, the values of $\delta^{13}C_{DIC}$ are low in May immediately after the

pulse flood event, comparable to the through-flow basin. However, following the flood event, the pulse-flooded basin takes on $\delta^{13}C_{DIC}$ characteristics of the non-flooded basins. This includes depressed values of $\delta^{13}C_{DIC}$ in July due to chemically enhanced CO_2 invasion, supported by high pH, low DIC concentration, and high macrophyte biomass (119 g/m^3).

The hydrological differences among the basin types also affect the $\delta^{13}C$ values of phytoplankton, which represents part of the base of the food-web. In May, $\delta^{13}C_{phytoplankton}$ is very low in the non-flooded basins because high availability of DIC (Figure 9a) allows preferential fractionation on the order of 25-30 ‰ (Figure 9f), as is typically observed when carbon is not limiting (Keeley and Sandquist 1992). By July, $\delta^{13}C_{phytoplankton}$ increased by more than 10‰ due to higher carbon demand by primary producers, which reduced isotopic fractionation to values between 5-10‰ (Figure 9f). Bicarbonate uptake may have partially contribute to this ^{13}C enrichment, as bicarbonate is isotopically heavier than $CO_{2(aq)}$, and is available for biological uptake at these pH levels Carbonate contamination was not a factor in the $\delta^{13}C$-enriched values since acidified and non-acidified samples had relatively the same $\delta^{13}C$ values (Appendix 2). By August, $\delta^{13}C_{phytoplankton}$ decreases slightly due to reduced carbon demand and corresponding increase in carbon isotopic fractionation. In sharp contrast, $\delta^{13}C_{phytoplankton}$ is nearly constant for the through-flow basin, because river water supplies sufficient DIC to meet the demand of primary producers. Values of $\delta^{13}C_{phytoplankton}$ in the pulse-flooded basin cannot be distinguished from those in the non-flooded basins, perhaps because growth of phytoplankton was delayed by high turbidity (Figures 8s, 9b). Low concentrations of DIC and low carbon demand by primary producers in May in the pulse-flooded basin combined to generate $\delta^{13}C_{phytoplankton}$ values that were similar to the $\delta^{13}C_{phytoplankton}$ in the non-flooded basins. By July, carbon demand by primary producers

(mainly macrophytes) led to carbon-limited growth that reduced isotopic fractionation, similar to the non-flooded basins. In August, the greater decline in $\delta^{13}C_{phytoplankton}$ in the pulse-flooded basin compared to non-flooded basins suggests a greater reduction in carbon demand. These results indicate that the $\delta^{13}C_{phytoplankton}$ can have a wide range of values over the course of the open-water season within an aquatic system and also among basins depending upon hydrological conditions, and carbon demand and availability. This finding suggests that studies of the role of flooding on aquatic food-webs cannot rely on a single point-in-time sampling, and instead must employ repeated sampling over the course of the open-water season.

Carbon and nitrogen isotope composition of the aquatic food-webs

Measurement of carbon and nitrogen isotope composition offers the opportunity to observe food-web ecology, assess trophic relationships and identify food-web responses to environmental attributes over ecologically-relevant time-scales. This is possible since the isotopic composition of an organism's tissue is related to that of the food it has assimilated as the tissue was generated (Vander Zanden and Rasmussen 1999). Carbon isotope composition is often used to indicate the food source as there is minimal fractionation of the two stable carbon isotopes (^{13}C and ^{12}C) between successive trophic levels. For example, the $\delta^{13}C$ of animals generally increases by 1 ‰ relative to their food source (DeNiro and Epstein 1978; Hecky and Hesslein, 1995). In contrast, stable nitrogen isotope composition is used to indicate the trophic position of organisms, because it is widely perceived that the $\delta^{15}N$ value tends to increase by approximately 3.4 ‰ between successive trophic levels (Vander Zanden and Rasmussen 1999). This enrichment occurs because the lighter ^{14}N is preferentially

excreted as a by-product of protein synthesis (DeNiro and Epstein 1981; Minagawa and Wada 1984; Kling et al. 1992). Top predators therefore should have the highest concentrations of ^{15}N. Interpreting the trophic links from the isotopic compositions of biota, however, is often fraught with challenges due to such influences as: organisms feed on multiple possible dietary sources in the natural environment, seasonal fluctuations in stable isotopic compositions of food sources due to variations in nutrient cycles, nutrient limitation and organism growth cycle, and modification of food sources by microbial processes (Minagawa and Wada 1984; Fry 2006; McGoldrick et al. 2008; O'Reilly and Hecky 2002). These factors contribute to the variability of $\delta^{13}C$ and $\delta^{15}N$ commonly observed in food-webs, and especially at the lower trophic levels of aquatic systems where the tissue turnover rate is higher than upper trophic levels and protein available for tissue assimilation in plants is lower in quality than in an animal (VanderZanden and Rassmusen 2001).

In the PAD study basins, biota in the lower trophic levels exhibited seasonal patterns of carbon isotope values that tracked those of the primary producers (their main food source) in the different hydrological basin types. In May, the consumers that occupy the lower trophic levels (i.e., pelagic grazers, epiphyte grazers, omnivores) had the most lowest seasonal $\delta^{13}C$ values, which corresponded to the primary producers under non-carbon limiting conditions. An increase in the $\delta^{13}C$ values occurred for these trophic levels between May and July due to consumption of isotopically heavier algae. By August, the decrease in $\delta^{13}C$ values of primary consumers reflected the reduction in $\delta^{13}C$ of the algae. In the through-flow basin, the $\delta^{13}C$ values of the lower trophic levels remained relatively constant throughout the open-water season since there was little fluctuation in $\delta^{13}C$ of the algae. The changes in $\delta^{13}C$ composition of the lower trophic levels in the pulse-flooded basin, however,

followed a similar seasonal pattern to that of the non-flooded basins. Thus, pulse floods do not appear to change carbon dynamics in the aquatic food-webs except to decrease early season concentration of DIC, which increases the potential for carbon limitation throughout the remainder of the open-water season.

The upper trophic levels (primary and secondary predators) did not experience similar seasonal patterns in $\delta^{13}C$ change seen in the lower trophic levels within each of the hydrologic basin types, suggesting that flooding exerts the strongest influence on the biota near the base of the aquatic food-webs (Figure 10). Differential effects of flooding on biota at lower and upper trophic levels may reflect that organisms at higher trophic levels are not strongly affected by flooding, or that the effects of flooding on C and N isotopic signatures are affected strongly by differences among organisms in their tissue turnover rates. Tissue turnover rate refers to the time required for the isotopic composition of an organism's tissue to be completely regenerated. Smaller, shorter-lived organisms that occupy the base of food-webs have more rapid tissue turnover rates, which enable the $\delta^{13}C$ of their tissues to more rapidly reflect changes in their nutrient source compared to larger, longer-lived organisms located at the upper trophic levels. For example, algae only require a few days for their tissue to fully turn over, whereas zooplankton can require over a month (Hecky 1991; Hyvonen 1997). Thus, each successive trophic level requires a longer period for the isotopic composition of tissue to represent that of the food it assimilated. Given that flooding exerts strong influence on physical and chemical conditions of pulse-flooded and through-flow basins that persists throughout the open-water season, we suggest that long tissue turnover times are the main factor that can account for the dampened seasonal fluctuations in carbon stable isotope compositions of primary predators and secondary predators relative to their

main food source and for similar carbon and nitrogen stable isotopic signatures in upper trophic levels of the different hydrological basin types.

A common paradigm in stable-isotope based food-web studies is that nitrogen enriches by approximately 3.4 ‰ between successive trophic levels due to loss of ^{14}N during protein synthesis (Minigawa and Wada 1984; Vander Zanden and Rassmusen 2001). In this study, however, no systematic enrichment was observed between successive trophic levels. Reasons to account for this discrepancy remain unknown, but one possibility is that the food-webs did not include fish. The average enrichment value of 3.4 ‰ was determined by comparing transfers between carnivorous biota and may not be accurate for trophic transfers between lower trophic levels and shorter-lived organisms (Minigawa and Wada 1984). For example, when omnivorous or herbivorous organisms were included in determining whole food-chain fractionation, the fractionation value (reported as the mean ± 1 standard deviation) of $\delta^{15}N$ is 3.4 ‰ ± 0.28 ‰ per trophic link between carnivores changed to 2.5 ‰ ± 2.5 ‰ per link between plants and herbivores (Vander Zanden and Rassmusen 2001). This difference in nitrogen isotope fractionation is related to the specialization in a consumer's diet (i.e., large if a strict predator, intermediate if omnivorous and small if a strict herbivore). Increased fractionation between carnivore trophic groups occurs because protein from prey is more abundant and easier to assimilate compared to plants, which have variable protein content and can be difficult to assimilate (Kling et al. 1992; Vander Zanden and Rassmusen 2001). The absence of clearly defined trophic nitrogen isotope fractionation in the observed PAD basins may, therefore, be due to the food-web being composed mainly of herbivores and omnivorous organisms.

Nitrogen cycling in these basins is not fully understood, but patterns of variation in $\delta^{15}N$ of food-web components did not differ between non-flooded and pulse-flooded basins, suggesting that pulse flood events do not strongly affect nitrogen dynamics relative to non-flooded basins. There was a slight effect of flooding on the food-web in the through-flow basin as multiple trophic groups had higher $\delta^{15}N$ values than in the other basins, suggesting that the invertebrates were consuming a slightly heavier $\delta^{15}N$ food source than the biota in the other basins. The secondary predators were consuming a mixture of algae and small organisms since they had similar $\delta^{15}N$ values to those of lower trophic groups, a finding that is consistent with observations of Matthews and Mazumder (2003) in four British Columbia lakes. In comparison, biota in the pulse-flooded basin tended to have $\delta^{15}N$ values similar to those in non-flooded basins, which tended to decrease between May and June and then increase in August.

Pulse flooding has short-lived effects on the aquatic food-webs in the PAD but longer effects on the limnological conditions. The $\delta^{13}C_{DIC}$ was initially low due to the river flooding, but shortly after the river disconnected from the basin, the $\delta^{13}C_{DIC}$ seasonal patterns mimicked those of the non-flooded basins. In these basins, photosynthesis by primary producers led to elevated pH, and decline of DIC concentration that led to atmospheric CO_2 invasion. The manipulation of the carbon pool by the primary producers was reflected in the isotopic composition throughout the rest of the affected food-webs. The $\delta^{15}N$ values of the primary producers did not appear to undergo discernible changes during the open-water season. In contrast, the majority of the ions maintained diluted concentrations similar to the adjacent river and may take several years for limnological conditions of pulse-flooded basins to acquire values typical of non-flooded basins due to slow working processes of evaporation

concentration of ions and dissolved nutrients as observed in PAD basins by Wiklund et al. (in preparation). This current study demonstrates that it is important to consider the frequency of sampling and time of year when sampling is conducted to assess the influence of short-lived events on limnological conditions and food-web dynamics of highly dynamic systems such as shallow, productive floodplain basins. Because there is marked seasonal variability of $\delta^{13}C_{DIC}$ and food-webs, a single sampling period in May, July, or August alone would not have allowed us to identify the variability in the nutrient dynamics, and single point-in-time sampling would have produced very different interpretations. Multiple sampling periods, therefore, are important to provide a means of tracking energy flow and cycles within rapidly fluctuating aquatic systems. The use of longer-lived primary consumers (e.g., mussels) to compare isotopic changes in higher trophic levels has been suggested by Cabana and Rasmussen (1996) and Post (2002) as a method to reduce temporal variance in the isotopic composition. However, this study shows that markedly shifting seasonal pattern in DIC concentration and $\delta^{13}C_{DIC}$ values were important to characterize in order to interpret food-web systematics.

Conclusions and Recommendations

The Peace-Athabasca Delta has experienced a long history of natural fluctuations in flooding regimes (e.g. Hugenholtz et al 2009; Wolfe et al. 2008b). While there have been several decades of studies to assess hydrological variability of this delta, the role of flooding on the ecology of floodplain basins has remained poorly understood. A common perception is that flooding provides a pulse of water and nutrients that maintain water levels and stimulate primary productivity (Prowse and Conly 1998), but there have been few direct measurements on the aquatic biota to confirm the ecological effects of floods in the PAD. Here, I show that flooding exerted strong control on basin hydrology and the physical and chemical conditions of PAD floodplain basins. Interestingly, pulse flood events had relatively short-lived effects on basin hydrology but longer lasting effects on limnological conditions that extended throughout the entire open-water season. For example, patterns of increasing $\delta^{18}O$ and declining water levels in the pulse-flooded basin were comparable to those of the non-flooded basins within a month after flooding, where evaporation dominated the water balance, but patterns of water chemistry remained different throughout the open-water season characterized by lower alkalinity and lower concentrations of DIC, DOC, chlorophyll *a*, and most ions except for SO_4 in the pulse-flooded basin. Although pulse flooding exerts strong influence on water chemistry conditions, it does not alter carbon dynamics, because seasonal patterns of variation in $\delta^{13}C_{DIC}$ values did not differ appreciably between the non-flooded and pulse-flooded basins but they differed from the pattern of the through-flow basin. Relatively constant $\delta^{13}C_{DIC}$ occurred in the through-flow basin due to supply of DIC from river water, whereas $\delta^{13}C_{DIC}$ decreased in July in the other basins. This pattern of decreasing $\delta^{13}C_{DIC}$ in the non-flooded and pulse-flooded basins is opposite of what would be expected during

conditions of high primary productivity and non-limiting carbon conditions, and occurred as photosynthesis increased pH and created non-equilibrium conditions that promoted chemical enhancement of atmospheric CO_2 invasion into the water, a process previously described by Herczeg and Fairbanks (1987). Fractionation of carbon by phytoplankton declined in mid-summer in the non-flooded and pulse-flooded basins due to carbon-limiting conditions, which did not occur in the through-flow basin. This hydrological control of $\delta^{13}C$ of DIC and phytoplankton was also reflected in the lower trophic levels (to primary consumers), but not in the upper trophic levels (primary and secondary predators). Marked differences between the through-flow basin and the other hydrological basin types (pulse- and non-flooded) in the $\delta^{13}C$ compositions of DIC and biota of lower trophic levels suggest that continuous river connection is required to modify carbon cycling within aquatic food-webs of the PAD and that the role of pulse flood events on some ecological processes is less influential then previously thought. However, further research should assess the role of flooding on the abundance and community composition of aquatic ecosystems.

Future studies in other shallow floodplain systems need to consider implementing both limnological and isotopic analysis in order to fully understand seasonal controls on the carbon and nitrogen isotope composition within the lower trophic levels of food-webs. The combined use of hydrolimnological and isotope tools to monitor seasonal changes across a gradient of flooding in the shallow basins of the PAD was necessary to identify seasonal carbon behavior. Multiple sampling periods during the open-water season was also essential in order to track short-lived fluctuations of nutrients in systems that experience flooding. It would have been difficult to correctly identify the controlling factors and response of the lower trophic levels if we relied on a single point-in-time sampling and did not include water

chemistry with isotope analysis. For example, Koster et al. (2008) attributed carbonate contamination as the source of enriched carbon isotope values for epiphytes, however multiple sampling throughout the season identified the enriched carbon isotopic composition was due to chemical enhancement of atmospheric CO_2 invasion and strong kinetic fractionation.

This study identified distinguishing $\delta^{13}C$ values of primary producers between the hydrological basin types, which provides new insight for interpreting $\delta^{13}C$ values of organic matter in sediment cores. The $\delta^{13}C$ records preserved in lacustrine sediments is often interpreted to represent variations in the photosynthesis-respiration cycle and indicate aquatic productivity. However, the non-flooded and pulse-flooded basins in the PAD experienced carbon limiting conditions which influenced the $\delta^{13}C$ composition of the lowest tropic levels in opposite directions to traditional interpretation of $\delta^{13}C$ signatures of organic matter, a finding that enables paleolimnologists to reconstruct past changes in flooding from sedimentary $\delta^{13}C$. Sampling of $\delta^{13}C$ of DIC and phytoplankton was key to develop this ability.

A complete understanding of the influence of flooding on the aquatic food-webs within the observed basins was not formed due to inadequate sample collection. It was difficult to predict how much sample was required to obtain $\delta^{15}N_{DIN}$ values of basin water. Water samples greater than 150 mL are required for $\delta^{15}N_{DIN}$ analysis in the PAD basins due to diluted concentrations of nitrogen in the basins. The nitrogen dynamics still remain difficult to interpret due to lack of understanding of fractionation between DIN and primary producers as well as variability in isotopic composition within the biota. Secondly, it would be valuable to incorporate information of the abundance, composition, and diversity of

aquatic biota between the hydrological basin types since they probably experience the greatest impact from flooding. I would also recommend an increase the sample size to enable statistical analysis to verify if there is a significant difference in the food-web communities with respect to flooding regimes. I initially intended to provide statistical analysis on community abundance and composition but there were few organisms represented in the 10 L samples collected in the basins.

Findings of this study have implications for natural resource managers of the PAD and other northern floodplain landscapes. Periodic short-lived flood events have long been widely believed to play an essential role in maintaining water levels and productivity of floodplain basins. Here, however, we show that seasonal patterns of change in water levels and carbon dynamics of non-flooded and pulse-flooded basins do not differ markedly. This finding demonstrates that pulse floods exert relatively short-lived effects on receiving basins which have been overestimated by past studies. Indeed, evidence by McGowan et al. (in review) and Wiklund (unpublished data) identifies that primary production is considerably higher in basins that have not flooded in two or more decades compared to basins that receive pulse floods in spring. As I show here, the pulse-flooded basin has lower concentrations of dissolved nutrients compared to the non-flooded basins. Thus, pulse flood events do not appear to elevate nutrients, nor do they stimulate primary production or alter carbon dynamics appreciably. Instead, regional climate conditions during the winter and open-water season are likely to play a more influential role on aquatic ecosystems than has been appreciated thus far. Snowmelt contributions from the local catchment as well as temperature and relative humidity during the growth season exert strong control on water levels of delta basins (Wolfe et al. 2007, 2008a). Based on the findings presented in this

thesis, the influence of variations in climate on aquatic productivity and food-web dynamics has been undervalued.

References

Allen, E.D. and D.H.N. Spence. 1981. The differential ability of aquatic plants to utilize the inorganic carbon supply in fresh waters. *New Phytologist* **87**: 269-283.

APHA. 2000. *Standard Methods for the Examination of Water and Wastewater*. 21st ed. Eaton, A.D., L.C. Clesceri, and A.E. Greenberg (eds.). American Public Health Association, Washington, DC.

Cabana, G. and J.B. Rasmussen. 1996. Comparison of aquatic food chains using nitrogen isotopes. *Proceedings of the National Academy of Sciences of the United States of America* **93**: 10844-10847.

Calder, J.A. and P.L. Parker. 1973. Geochemical implications of induced changes in ^{13}C fractionation by blue-green algae. *Geochimica et Cosmochimica Acta* **37**: 133-140.

Coplen, T.B. 1996. New guidelines for reporting stable hydrogen, carbon, and oxygen isotope-ratio data. *Geochimica et Cosmochimica Acta* **60**: 3359-3360.

Dean, W.E. and M. Stuiver. 1993. Stable carbon and oxygen isotope studies of the sediments of Elk Lake, Minnesota. In Bradbury J.P. and W.E. Dean(eds.), *Elk Lake, Minnesota: Evidence for rapid climate change in the North-Central United States.* Geological Society of America. Special Paper 276, Boulder (CO), pp. 163-180.

del Giorgio, P.A. and R.L. France. 1996. Ecosystem-specific patterns in the relationship between zooplankton and POM or microplankton $\delta^{13}C$. *Limnology and Oceanography* **41**(2):359-365.

DeNiro, M.J. and S. Epstein. 1981. Influence of diet on the distribution of carbon isotopes in animals. *Geochimica et Cosmochimica Acta* **45**:341-351.

DeNiro, M.J. and S. Epstein. 1978. Influence of diet on the distribution of nitrogen isotopes in animals. *Geochimica et Cosmochimica Acta* **42**:495-506.

Deuser W.G., E.T. Degens, and R.R.L. Guillard. 1968. Carbon isotope relationships between plankton and sea-water. *Geochimica et Cosmochimica Acta* **32**:657-660.

Drimmie, R.J. and R.A. Heemskerk. 1993. Water ^{18}O by CO_2 Equilibration. Technical Procedure 13.0, Rev. 02. Environmental Isotope Laboratory, Department of Earth Sciences, University of Waterloo.

Duthie, H.C., J.-R. Yang, T.W.D. Edwards, B.B. Wolfe, and B.G. Warner. 1996. Hamilton Harbour, Ontario: 8300 years of limnological and environmental change inferred from microfossil and isotopic analyses. *Journal of Paleolimnology* **15**: 79-97.

Emrich, K., D.H. Ehhalt, and J.C. Vogel. 1970. Carbon isotope fractionation during the precipitation of calcium carbonate. *Earth and Planetary Science Letters* **8**: 363-371.

Environment Canada. 1996. Manual of analytical methods. The National Laboratory of Environmental Testing, Canada Centre for Inland Waters, Burlington.

Feuchtmayr, H. and J. Grey. 2003. Effect of preparation and preservation procedures on carbon and nitrogen stable isotope determinations from zooplankton. *Rapid Communications in Mass Spectrometry* **17**:2650-2610.

Fry, B. 2006. *Stable Isotope Ecology*. Springer, New York. 308p.

Furgal, J. and R. Smith. 1997. Ultraviolet radiation and photosynthesis by Georgian Bay phytoplankton of varying nutrient and photoadaptive status. *Canadian Journal of Fisheries and Aquatic Sciences* **54**:1659-1667.

Hall, R.I., B.B. Wolfe, and T.W.D. Edwards, 2004. *A Multi-century Flood, Climatic, and Ecological History of the Peace-Athabasca Delta, Northern Alberta, Canada*. Final Report. Published by BC Hydro. 163p + Appendices.

Hecky, R.E., 1991. The pelagic ecosystem, p. 90-110. *In* G.W. Coulter [ed.], Lake Tanganyika and its Life. Oxford University Press.

Hecky, R.E. and R.H. Hesslein. 1995. Contributions of benthic algae to lake food webs as revealed by stable isotope analysis. *Journal of the North American Benthological Society* **14**: 631-653.

Herczeg, A.L. and R.G. Fairbanks. 1987. Anomalous carbon isotope fractionation between atmospheric CO_2 and dissolved inorganic carbon induced by intense phytosynthesis. *Geochimica et Cosmochimica Acta* **51**: 895-899.

Hugenholtz, C.H., D.G. Smith, and J. M. Livingston. 2009. Application of floodplain stratigraphy to determine the recurrence of ice-jam flooding along the lower Peace and Athabasca Rivers, Alberta. *Canadian Water Resources Journal* **34**: 1-16.

Hyvonen, K. 1997. Study of zooplankton development time at Lake Tanganyika. Finnish International Development Agency/Food and Agriculture Organization of the United Nations, GCP/RAF/217/FIN-TD/78 (En).

Junk, W.J. 2005. Flood pulsing and the linkages between terrestrial, aquatic, and wetland systems. *Verhandlungen-Internationale Vereinigung Limnol*ogie **29**:11-38.

Keeley, J.E., and D.R. Sandquist. 1992. Carbon: Freshwater plants. *Plant, Cell and Environment* **15**: 1021-1035.

Kling, G.W., B. Fry, and W.J. O'Brien. 1992. Stable isotopes and planktonic trophic structure in Arctic lakes. *Ecology* **73**: 561-566.

Köster, D., R.I. Hall, and J. Marty. 2008. Influence of floods on the food-web structure of two freshwater delta lakes as determined by carbon and nitrogen stable isotopes. *Verhandlungen-Internationale Vereinigung Limnol*ogie. **30**: 497-500.

Lesack, L., P. Marsh, and R.E. Hecky. 1998. Spatial and temporal dynamics of major solute chemistry among Mackenzie Delta lakes. *Limnology and Oceanography* **43**: 1530-1543.

Lewis, W.M. Jr., S.K. Hamilton, M.A. Lasi, M. Rodríguez, and J.F. Saunders III. 2000. Ecological determinism on the Orinoco floodplain. *BioScience* **50**: 681-692.

Lindholm, M., D.O. Hessen, K. Mosepele, and P. Wolski. 2007. Food webs and energy fluxes on a seasonal floodplain: the influence of flood size. *Wetlands* **27**: 775-784.

Matthews, B. and A. Mazumder. Compositional and interlake variability of zooplankton affect baseline stable isotope signatures. *Limnology and Oceanography* **48**: 1977-1987.

McGowan, S.M., P.R. Leavitt, R.I. Hall, B.B. Wolfe, T.W.D. Edwards, T. Karst-Riddoch, and S.R. Vardy. In Review. Declining flood frequency increases lake primary production in a northern river delta.

McKenzie, J.A. Carbon isotopes and productivity in the lacustrine and marine environment. Chemical Processes in Lakes, John Wiley and Sons, New York New York. 1985.

McGoldrick, D.J., D.R. Barton, M. Power, R.W. Scott, and B.J. Butler. 2005. Dynamics of bacteria-substrate stable isotope separation: dependence on substrate availability and implications for aquatic food web studies. *Canadian Journal of Fisheries and Aquatic Sciences* **65**: 1983-1990.

Merritt, R.W. and Cummins, K.W. 1996. *An Introduction to the Aquatic Insects of North America* (3rd Ed.). Kendall/Hunt Publishing Company, Dubuque.

Minagawa, M. and E. Wada. 1984. Stepwise enrichment of ^{15}N along food chains: Further evidence and the relation between $\delta^{15}N$ and animal age. *Geochimica et Cosmochimica Acta* **48**: 1135-1140.

Morrison, J., Brockwell, T., Merren, T., Fourel, F., Phillips, A.M., 2001. A new on-line method for high precision stable-hydrogen isotopic analyses on nanolitre water samples. *Analytical Chemistry* **73**: 3570–3575.

O'Leary, M.H. 1981. Carbon isotope fractionation in plants. *Phytochemistry* **20**: 553-567.

O'Reilly, C.M. and R.E. Hecky. 2002. Interpreting stable isotopes in food webs: recognizing the role of time averaging at different trophic levels. *Limnology and Oceanography* **47**: 306-309.

Peace-Athabasca Delta Implementation Committee (PAD-IC). 1987. *Peace-Athabasca Delta Water Management Works Evaluation - Final Report, including Appendices*. Governments of Canada, Saskatchewan and Alberta. 63p.

Peace-Athabasca Delta Project Group (PAD-PG). 1973. *Peace-Athabasca Delta Project Group Technical Report: a Report on Low Water Levels in Lake Athabasca and their Effect on the Peace-Athabasca Delta.* Governments of Canada: Alberta and Saskatchewan. 176p.

Peace-Athabasca Delta Technical Studies (PAD-TS). 1996. *Final Report.* PADTS Steering Committee, Fort Chipewyan, Alberta. 88p.

Peters, D.L., T.D. Prowse, A. Pietroniro, and R. Leconte. 2006. Flood hydrology of the Peace-Athabasca Delta, northern Canada. *Hydrological Processes* **20**: 4073-4096.

Pinnegar, J.K. and N.V.C. Polunin. 1999. Differential fractionation of $\delta^{13}C$ and $\delta^{15}N$ among fish tissues: implications for the study of trophic interactions. *Functional Ecology* **13**: 225-231.

Prowse, T.D. and F.M. Conly. 1998. Effects of climatic variability and flow regulation on ice-jam flooding of a northern delta. *Hydrological Processes* **12**: 1589-1610.

Prowse, T.D. and M.N. Demuth. 1996. Using ice to flood the Peace-Athabasca Delta, Canada. *Regulated Rivers: Research & Management* **12**: 447-457.

Post, D.M. 2002. Using stable isotopes to estimate trophic position: models, methods, and assumptions. *Ecology* **83**: 703-718.

Quay, P.D., S.R. Emerson, B.M. Quay, and A.H. Devol. 1986. Ther carbon cycle for Lake Washington – a stable isotope study. *Limnology and Oceanography* **31**: 596-611.

Rand, M.C., A.E. Greenberg, M.J. Taras. 1976. Standard methods for the Examination of Water and Wastewater. 14th Edition. USA. APHA-AWWA-WPCF.

Rau, G. 1978. Carbon-13 depletion in a subalpine lake: Carbon flow implications. *Science* **201**:901-902.

Roozen, F.C.J.M. 2005. Transparency of floodplain lakes; a study of plankton and suspended matter along the lower Rhine. Doctoral thesis. Wageningen Universiteit.

Sarakinos, H.C., M.L. Johnson and M.J. VanderZanden. 2002. A synthesis of tissue-preservation effects on carbon and nitrogen stable isotope signatures. *Canadian Journal of Zoology* **80**:381-387.

Schelske, C.L. and D.A. Hodell. 1995. Using carbon isotopes of bulk sedimentary organic matter to reconstruct the history of nutrient load. *Limnology and Oceanography* **40**: 918-929.

Schelske, C.L. and D.A. Hodell. 1991. Recent changes in productivity and climate of Lake Ontario detected by isotopic analysis of sediment. *Limnology and Oceanography* **36**: 961-975.

Schindler, D. W., and W. F. Donahue. 2006. An impending water crisis in Canada's western prairie provinces, Proceedings of the National Academy of Sciences of the United States of America **103**: 7210–7216.

Schindler, D.W. and J.P. Smol. 2006. Cumulative effects of climate warming and other human activities on freshwaters of arctic and subarctic North America. *Ambio* **35**: 160-168.

Sierszen, M.E., G.S. Peterson, A.S. Trebitz, J.C. Brazner, and C.W. West. 2006. Hydrology and nutrient effects on food-web structure in ten Lake Superior coastal wetlands. *Wetlands* **26**: 951-964.

Solomon, C.T., S.R. Carpenter, J.J. Cole, and M.L Pace. 2008. Support of benthic invertebrates by detrital resources and current autochthonous primary production: results from a whole-lake ^{13}C addition. *Freshwater Biology* **53**: 42-54.

Squires, M.M. and L.F.W. Lesack. 2002. Water transparency and nutrients as controls on phytoplankton along a flood-frequency gradient among lakes of the Mackenzie Delta, western Canadian Arctic. Canadian *Journal of Fisheries and Aquatic Science* **59**:1339-1349.

Takahashi, K., T. Yoshioka, E. Wada, and M. Sakamoto, 1990. Temporal variations in carbon isotope ratio of phytoplankton in a eutrophic lake. *Journal of Plankton Research*. **12**: 799-808

ter Braak C.J.F. and P. Šmilauer. 1998. *CANOCO for Windows (v. 4.0).* Centre for Biometry, Wageningen, The Netherlands.

Tockner, K., D. Pennetzdorfer, N. Reiner, F. Schiemer, and J.V. Ward. 1999. Hydrological connectivity, and the exchange of organic matter and nutrients in a dynamic river-floodplain system (Danube, Austria). *Freshwater Biology* **41**: 521-535.

Townsend, G.H. 1975. Impact of the Bennett Dam on the Peace-Athabasca Delta. *Journal of the Fisheries Research Board of Canada* **32**: 171-176.

Vander Zanden, M.J. and J.B. Rasmussen. 1999. Primary consumer $\delta^{13}C$ and $\delta^{15}N$ and the trophic position of aquatic consumers. *Ecology* **80**: 1395-1404.

Vander Zanden, M.J. and J.B. Rasmussen. 2001. Variation in $\delta^{15}N$ and $\delta^{13}C$ trophic fractionation: Implications for aquatic food web studies. *Limnology and Oceanography*. **46**: 2061-2066.

Wachniew, P. and K. Rozanski. 1997. Carbon budget of a mid-latitude, groundwater-controlled lake: isotopic evidence for the importance of dissolved inorganic carbon recycling. *Geochimica et Cosmochimica Acta* **61**: 2453-2465.

Wantzen K.M., W.J. Junk and K-O. Rothhaupt. 2008. An extension of the floodpulse concept (FPC) for lakes. *Hydrobiologia* **613**: 151-170.

Wetzel, R.G. 2001. Limnology: lake and river ecosystems. 3rd Ed. Academic Press. New York New York.

Wetzel, R.G. 1995. Death, detritus, and energy flow in aquatic ecosystems. *Freshwater Biology* **33**: 83-89.

Whitmore, T.J, M. Brenner, Z. Jiang, J.H. Curtis, A.M. Moore, D.R. Engstrom, and Y. Wu. 1997. Water quality and sediment geochemistry in lakes of Yunnan Province, southern China. *Environmental Geology and Water Sciences* **32**: 45-55.

Wiklund, J.A. Unpublished data. Macrophyte communities of the Peace-Athabasca Delta: Hydro-limnologic control of community composition and light limitation of community biomass. PhD Thesis in preparation. Chapter 4.

Wiklund, J.A., N.B. Bozinovski, R.I. Hall, and B.B. Wolfe. 2010. Epiphytic diatoms as flood indicators. *Journal of Paleolimnology* **44**: 25-42.

Wolfe, B.B., R.I. Hall, T.W.D. Edwards, S.R. Vardy, M.D. Falcone, C. Sjunneskog, F. Sylvestre, S. McGowan, P.R. Leavitt and P. van Driel. 2008a. Hydroecological responses of the Athabasca Delta, Canada, to changes in river flow and climate during the 20th century. *Ecohydrology* **1**:131-148.

Wolfe, B.B., R.I. Hall, T.W.D. Edwards, S.R. Jarvis, R.N. Sinnatamby, Y. Yi and J.W. Johnston. 2008b. Climate-driven shifts in quantity and seasonality of river discharge over the past 1000 years from the hydrographic apex of North America. *Geophysical Research Letters* **35**: L24402.

Wolfe, B.B., T.L. Karst-Riddoch, R.I. Hall, T.W.D. Edwards, M.C. English, R. Palmini, S. McGowan, P.R. Leavitt and S.R. Vardy. 2007. Classification of hydrologic regimes of northern floodplain basins (Peace-Athabasca Delta, Canada) from analysis of stable isotopes ($\delta^{18}O$, $\delta^{2}H$) and water chemistry. *Hydrological Processes* **21**:151-168.

Wolfe, B.B., T.L. Karst-Riddoch, S.R. Vardy, M.D. Falcone, R.I. Hall, and T.W.D. Edwards. 2005. Impacts of climate and river flooding on the hydro-ecology of a floodplain basin, Peace-Athabasca Delta, Canada since A.D. 1700. *Quaternary Research* **64**:147-162.

Wolfe, B.B., T.W.D. Edwards, R. Aravena, and G.M. Macdonald. 1996. Rapid Holocene hydrologic change along boreal tree-line revealed by $\delta^{13}C$ and $\delta^{18}O$ in organic lake sediments, Northwest Territories, Canada. *Journal of Paleolimnology* **15**: 171-181.

Wood Buffalo National Park. 2007. Wood Buffalo National Park flood monitoring map.

Wood Buffalo National Park. 2008. Wood Buffalo National Park flood monitoring map.

Appendix 1

Average daily water level logger data used for hydrological analysis of the four study basins. The water levels are in centimetres.

	May				June				July				August			
Day	**PAD 1**	**PAD3**	**PAD31**	**PAD45**	**PAD 1**	**PAD3**	**PAD31**	**PAD45**	**PAD 1**	**PAD3**	**PAD31**	**PAD45**	**PAD 1**	**PAD3**	**PAD31**	**PAD45**
1	-	-	-	-	106.49	70.70	115.44	137.38	96.24	59.54	106.40	138.74	88.78	52.93	95.32	130.75
2	-	-	-	-	106.60	70.70	113.92	136.60	95.38	58.62	105.20	139.80	88.35	52.62	94.56	125.87
3	-	-	-	-	107.16	71.21	113.61	137.11	95.16	58.36	104.96	141.17	86.97	51.43	94.00	121.88
4	-	-	-	-	107.31	71.49	113.11	137.28	95.13	58.32	104.72	142.30	87.08	51.12	93.78	123.03
5	-	-	-	-	107.40	71.51	112.83	141.15	94.42	57.63	103.67	139.97	87.32	51.42	93.96	123.05
6	-	-	-	-	107.15	71.59	112.68	142.35	95.16	58.53	104.37	142.89	87.24	51.38	93.88	121.23
7	-	-	-	-	106.49	71.16	111.92	141.96	96.02	59.42	105.47	144.78	87.64	52.04	94.21	119.58
8	-	-	-	-	107.43	71.92	112.51	144.20	95.30	59.09	104.44	141.69	89.31	54.19	95.61	124.30
9	-	-	-	-	106.49	71.25	111.71	141.18	95.66	59.11	104.35	141.24	89.21	54.20	95.14	121.10
10	-	-	-	-	104.93	69.61	110.12	139.66	94.58	58.23	103.84	141.67	88.46	53.25	94.56	118.60
11	-	-	-	-	104.66	68.92	109.76	144.76	94.53	57.62	102.93	138.83	88.82	53.38	94.56	117.38
12	-	-	-	-	104.88	69.06	114.13	148.86	92.83	55.85	101.34	138.06	88.24	52.99	93.99	115.76
13	-	-	-	-	104.04	68.30	125.59	150.43	92.24	55.23	100.92	138.02	87.92	52.63	93.81	115.81
14	-	-	-	-	103.48	67.82	140.77	149.19	93.47	56.46	101.34	137.73	88.31	52.88	93.75	115.54
15	-	-	-	-	103.49	67.74	148.14	152.32	92.49	55.39	100.64	136.13	88.06	52.62	93.38	115.29
16	-	-	-	-	102.97	67.32	142.17	152.22	91.44	54.66	99.97	135.53	87.56	52.13	92.74	112.27
17	-	-	-	-	101.87	66.05	133.01	148.84	90.90	53.87	99.19	134.82	86.98	51.61	92.16	109.37
18	-	-	-	-	102.12	66.18	126.59	145.27	90.54	53.70	98.82	134.01	85.90	50.70	91.13	107.15
19	-	-	-	-	102.19	66.08	122.22	144.97	90.89	54.15	98.81	132.37	85.51	50.28	90.31	105.09
20	-	-	-	-	102.01	65.81	118.94	143.54	90.82	54.49	98.24	131.59	86.02	50.82	90.79	105.69
21	-	-	-	-	101.61	65.05	116.38	142.04	90.80	54.41	98.12	131.90	87.66	55.21	93.36	114.23
22	-	-	-	-	100.03	63.69	113.65	140.89	90.02	53.57	97.50	132.60	-	60.39	97.84	122.39
23	112.60	75.00	-	-	99.73	62.95	112.08	139.65	89.09	52.51	96.46	132.22	-	60.78	97.81	116.65
24	111.90	75.03	-	-	99.12	62.40	110.69	138.69	90.10	53.54	97.23	133.55	-	60.44	97.49	-
25	110.67	74.15	-	120.00	98.28	61.43	109.58	135.86	91.29	55.52	98.29	134.95	-	60.51	-	-
26	109.36	72.90	-	120.84	98.28	61.41	109.19	137.35	90.88	54.77	97.63	133.17	-	59.39	-	-
27	108.91	72.51	127.00	122.54	97.06	60.27	107.85	136.68	90.06	54.09	96.76	130.07	-	60.85	-	-
28	111.28	74.98	128.91	146.20	96.05	59.16	106.69	135.75	89.22	53.71	95.83	129.48	-	-	-	-
29	110.31	74.28	125.57	145.09	95.56	58.35	105.94	133.90	89.19	53.68	95.61	129.20	-	-	-	-
30	107.66	71.63	120.87	138.98	96.33	59.21	106.47	136.60	89.02	53.51	95.69	128.65	-	-	-	-
31	106.72	70.68	117.60	138.15					89.00	53.32	95.78	129.13	-	-	-	-

Appendix 2

Water isotope values for 2007 and 2008 at the four basins and two river locations in the PAD. These values were used to identify the main hydrological controls on the water balance in the basins. Replicate analyses were conducted on the same sample as indicated by columns '1' and '2'.

Site ID	Sample Date	$\delta^{18}O$			$\delta^{2}H$		
	(mm/yy)	1	2	Avg.	1	2	Avg.
PAD 1	05/07	-11.48		-11.48	-114.48	-114.16	-114.32
	06/07	-10.10		-10.10	-109.01	-109.02	-109.02
	07/07	-8.27	-8.14	-8.21	-100.22	-99.76	-99.99
	08/07	-7.83		-7.83	-97.05	-97.09	-97.07
	09/07	-8.70		-8.70	-99.19	-99.38	-99.29
	10/07	-11.26		-11.26	-108.00	-108.48	-108.24
	05/08	-13.30		-13.30	-121.66	-121.53	-121.60
PAD 3	05/07	-12.16		-12.16	-119.69	-120.07	-119.88
	07/07	-7.40		-7.40	-96.18	-96.32	-96.25
	08/07	-7.73		-7.73	-96.66	-95.84	-96.25
	09/07	-9.97	-9.99	-9.98	-103.44	-102.87	-103.16
	10/07	-10.89		-10.89	-108.94	-108.47	-108.71
	05/08	-13.86	-13.78	-13.82	-125.90	-125.13	-125.52
PAD 31	05/07	-17.19	-17.01	-17.10	-136.20	-137.61	-136.91
	07/07	-14.98		-14.98	-129.14	-128.22	-128.68
	08/07	-13.41		-13.41	-120.20	-121.21	-120.70
	09/07	-12.94		-12.94	-119.22	-119.66	-119.44
	10/07	-13.25	-13.30	-13.27	-119.32	-119.54	-119.43
	05/08	-18.15		-18.15	-144.11	-144.74	-144.43
PAD 45	05/07	-17.31		-17.31	-137.87	-137.63	-137.75
	06/07	-18.83	-18.57	-18.70	-147.08	-147.09	-147.09
	07/07	-18.00		-18.00	-140.64	-140.99	-140.82
	08/07	-17.46		-17.46	-136.88	-136.70	-136.79
	08/07	-15.74		-15.74	-129.79	-129.58	-129.69
	10/07	-16.36		-16.36	-133.73	-132.99	-133.36
	05/08	-18.20		-18.20	-144.59	-144.57	-144.58
PAD R11	05/07	-17.71		-17.71	-140.86	-140.72	-140.79
	06/07	-17.94		-17.94	-143.26	-143.73	-143.50
	07/07	-16.76		-16.76	-137.16	-137.51	-137.34
	08/07	-16.11	-16.30	-16.21	-131.65	-131.44	-131.55
	09/07	-16.48		-16.48	-134.62	-134.60	-134.61
	10/07	-16.13		-16.13	-133.49	-133.42	-133.46
	05/08	-18.00		-18.00	-144.84	-145.26	-145.05
PAD RMAM	05/07	-17.25		-17.25	-139.45	-139.15	-139.30
	07/07	-17.94		-17.94	-142.08	-143.16	-142.62
	08/07	-17.20		-17.20	-136.10	-136.74	-136.42
	09/07	-16.82		-16.82	-135.82	-135.73	-135.78
	10/07	-16.45	-16.48	-16.46	-133.43	-133.18	-133.31
	05/08	-18.25	-18.04	-18.15	-144.78	-144.39	-144.59

Appendix 3

Concentrations of nutrients and ions in water samples from the study basins that were presented in Figures 7 and 8 to identify seasonal patterns of limnological condition in the four basins and the influence of river water on the through-flow and pulse-flooded basins. Questionable values are in bold and indicated with a '*' – these values were omitted from PCA analysis.

Site ID	Date Sampled (mm/yy)	TKN mg/L	NO_3NO_2 mg/L	NH_3 mg/L	TP mg/L	TDP mg/L	DOC mg/L	DIC mg/L	Cl mg/L	Ca mg/L	Mg mg/L	Na mg/L	K mg/L	SiO_2 mg/L	SO_4 mg/L
PAD1	05/07	1.77	0.012	0.182	0.0236	0.0562	36.5	39.7	11.5	39.7	15.9	15	9.14	1.58	5.95
	07/07	2.41	0	0.072	0.0258	0.0345	49.1	19	10.7	20.5	19.6	20.2	6.88	1.36	2.12
	08/07	2.09	0.019	0.142	0.0247	0.032	49.5	23.4	11.9	18.6	20	21.9	7.62	0.73	3.6
	05/08	**42***	0.007	**32***	0.0202	**8.65***	39.4	35.5	15.9	35.6	17.1	19.1	**17.2***	**5.05***	5.14
PAD3	05/07	1.54	0.007	0.134	0.018	0.0644	23.1	41.1	9.07	39.4	15.6	9.23	18.9	4.84	12
	07/07	2.2	0	0.071	0.0256	0.0388	33.5	25	8.76	15.4	18	14.2	21.3	2.83	4.18
	08/07	2.51	0.268	0	0.0277	0.0438	33.2	31.1	8.97	19.8	18.5	14.1	21.8	0.26	9.45
	05/08	2	0.014	0.055	0.021	0.0719	24.8	38.1	7.64	38.1	15.3	9.23	15.2	4.04	8.79
PAD31	05/07	0.696	0.014	0.073	0.0162	0.0451	17.6	22.2	5.35	29.3	6.37	9.97	1.94	5.82	16.5
	07/07	0.749	0	0.054	0.0395	0.0582	12.9	11.9	4.75	20.2	9.24	9.84	0.51	2.15	20
	08/07	0.841	0.154	0	0.0275	0.0414	14.4	16.7	5.25	19.1	9.89	**1.18***	**10.7***	0.17	18.4
	05/08	0.704	0.017	0.087	0.022	0.0507	15.9	20.5	5.02	28.1	6.37	10.5	2.05	4.68	16.8
PAD45	05/07	0.693	0.067	0.027	0.0223	0.103	17.1	20.3	5.47	27	5.91	9.92	1.83	5.71	16
	07/07	0.331	0	0.046	0.0288	0.0366	7.2	22.1	7.95	30	7.77	10.7	1.03	3.63	21.4
	08/07	0.352	0.081	0	0.0088	0.0301	9.4	22.7	8.68	30.4	7.99	**1.03***	**11.8***	4.46	22.4
	05/08	0.571	0.063	0.032	0.0182	0.113	15.2	17.3	5.01	24	5.84	10.3	1.56	6.09	15.2
PADR11	05/07	0.747	0.128	0.006	0.0223	0.207	17.7	18.7	3.91	27.3	5.46	8.19	2.04	5.88	14.8
	07/07	0.346	0.017	0.015	0.0082	0.0659	6.6	17.7	5.41	23.4	5.94	7.46	1.14	3.74	14.9
	08/07	0.294	0.068	0	0.0089	0.0394	6.3	14.4	5.63	19.5	5.05	7.07	1.11	4.07	13.3
	05/08	0.542	0.107	0.005	0.022	0.113	13.2	15.1	4.41	21.2	5	8.41	1.63	5	13.1
PADRMAM	05/07	0.691	0.039	0.027	0.0185	0.113	16.4	20.9	5.82	28.6	6.35	10.2	1.74	5.62	17.4
	07/07	0.325	0	0.016	0.0099	0.0514	6.7	21.1	6.4	29.1	7.57	9.74	0.96	3.64	21.8
	08/07	0.425	0.103	0	0.0127	0.05	10.5	23.3	10.2	31.4	8.39	**1.07***	**13.4***	5.14	22.4
	05/08	0.535	0.066	0.01	0.0158	0.176	15.4	17	5.07	24.2	5.82	10.2	1.62	6.09	15.4

Appendix 4

Water quality data used in Figures 7 and 8 to identify seasonal patterns within the four basins and two river sample locations. Questionable data are indicated with a '*' and were omitted from analysis.

Site ID	Date Sampled (mm/yy)	SPCOND uS/CM	Alk mg/L	TSS mg/L	ISS mg/L	OSS mg/L	Chlor a ug/L	k_dPAR umol photons $m^{-2}s^{-1}$
PAD1	05/07	376	173	5.20	1.10	4.10	10.97	1.80
	07/07	303	133	2.00	0.33	1.66	4.96	3.03
	08/07	300	142	2.45	0.65	1.80	5.87	2.31
	05/08	**620***	-	5.70	1.10	4.60	12.61	2.38
PAD3	05/07	387	178	7.88	1.44	6.44	8.18	1.86
	07/07	297	139	3.25	0.85	2.40	6.17	5.68
	08/07	328	149	4.86	2.22	2.64	4.24	2.13
	05/08	363	173	7.81	2.15	5.67	9.23	2.34
PAD31	05/07	230	91.8	20.03	15.78	4.25	4.97	4.01
	07/07	200	76.5	4.61	2.86	1.74	0.91	2.65
	08/07	208	81.3	3.50	1.45	2.05	4.23	1.70
	05/08	230	93	76.12	54.11	22.01	4.73	2.77
PAD45	05/07	216	84.7	81.67	73.75	7.92	2.81	7.84
	07/07	251	94.9	17.68	14.20	3.48	6.05	2.68
	08/07	256	98.2	17.70	14.90	2.80	5.59	2.71
	05/08	204	79.9	88.30	78.30	10.00	6.05	6.91
PADR11	05/07	205	81.6	215.00	190.83	24.17	2.23	17.22
	07/07	192	73.4	52.40	5.55	46.85	5.36	6.16
	08/07	170	62.8	35.41	4.75	30.67	15.93	4.15
	05/08	182	70.2	90.52	80.64	9.88	6.10	8.66
PADRMAM	05/07	226	89.1	145.22	131.21	14.01	5.60	9.63
	07/07	238	90.8	93.08	83.90	9.18	5.34	6.36
	08/07	271	101	54.33	47.12	7.22	9.28	3.99
	05/08	206	80.3	148.73	135.47	13.26	8.14	7.90

Appendix 5

Profiles of pH in water of the four study basins. The average values were used within the seasonal profiles of the basins in Figures 7, 8 and 9.

Site ID	Date Sampled	Depth (cm)												
	(mm/yy)	10	20	30	40	50	60	70	80	90	100	110	120	130
PAD1	05/07	7.07	7.21	7.36	7.44	7.5	7.58	7.63	7.67	7.7	7.73			
	07/07	9.88	9.91	9.72	9.53	9.32								
	08/07	9.69	9.69	9.7	9.7	9.7	9.71	9.71	9.66					
	05/08	7.42	7.44	7.38	7.35	7.33	7.31	7.3	7.3	7.29	7.29	7.29		
PAD 3	05/07	8.08	8.12	8.16	8.17	8.14	8.19	8.19						
	07/07	9.57	9.62	9.62	9.62	9.27								
	08/07	9.02	9.02	9.03	9.03	9.04								
	05/08	7.74	7.73	7.72	7.71	7.71	7.71	7.71	7.7					
PAD 31	05/07	7.64	7.63	7.62	7.62	7.62	7.63	7.62	7.62	7.62	7.61	7.59	7.54	
	07/07	10	9.96	9.89	9.85	9.83	9.83	9.85	9.83					
	08/07	9.36	9.36	9.36	9.36	9.36	9.37	9.38	9.39					
	05/08	7.91	7.86	7.88	7.85	7.86	7.86	7.85	7.83	7.82	7.83			
PAD 45	05/07	7.86	7.82	7.78	7.76	7.75	7.74	7.74	7.74	7.74	7.73	7.73	7.73	7.73
	07/07	7.32	7.43	7.51	7.58	7.6	7.62	7.62	7.66	7.68	7.68	7.7	7.72	
	08/07	8.2	8.16	8.15	8.15	8.16	8.16	8.16	8.15	8.14	8.15			
	05/08	8.03	7.96	7.94	7.9	7.86	7.86	7.82	7.81	7.78	7.78	7.76		

Appendix 6

Inorganic and organic carbon isotope composition of the four study basins and two rivers during the four sample periods. The DIC values were used to identify the seasonal carbon dynamics identified in the non-flooded and pulse-flooded basins (Figure 9). However, DOC isotope values were not used within the body of this thesis because they did not contribute to the analysis.

Site ID	Date Sampled (mm/yy)	$\delta DI^{13}C$ ‰	$\delta DO^{13}C$ ‰
PAD 1	05/07	-4.21	-26.30
	07/07	-11.96	-25.61
	08/07	-10.95	-25.81
	05/08	-4.28	-25.98
PAD 3	05/07	-5.17	-26.77
	07/07	-12.95	-26.61
	08/07	-10.45	-26.42
	05/08	-5.54	-27.20
PAD 31	05/07	-11.36	-26.99
	07/07	-16.11	-26.46
	08/07	-14.38	-26.12
	05/08	-10.10	-27.19
PAD 45	05/07	-11.02	-26.96
	07/07	-7.81	-26.80
	08/07	-7.46	-26.57
	05/08	-9.76	-26.87
PAD R11	05/07	-11.57	-27.03
	07/07	-7.61	-26.95
	08/07	-6.30	-26.63
	05/08	-9.78	-26.74
PAD RMAM	05/07	-10.69	-27.11
	07/07	-7.82	-26.68
	08/07	-7.82	-26.69
	05/08	-10.05	-26.82

Appendix 7

Aquatic invertebrate $\delta^{13}C$, $\delta^{15}N$, %C, %N, and C/N values in PAD 1 during each sampling campaign (month/year). Invertebrates were grouped according to feeding groups and averaged in order to track response to seasonal changes in nutrient availability. An organism with a 'B' at the end of their ID indicates that they were obtained in the benthic samples.

			05/07					07/07		
Sample ID	**δ13C**	**δ15N**	**%C**	**%N**	**C/N**	**δ13C**	**δ15N**	**%C**	**%N**	**C/N**
Bosmina						-20.78	2.07	37.78	6.23	6.07
Calanoida	-31.15	6.87	43.44	9.55	4.55					
Calanoida	-19.73	1.44	52.52	6.15	8.54					
Chaoborus	-21.22	4.40	53.03	11.39	4.66					
Cyclopoida						-18.71	1.74	45.53	6.47	7.03
Daphnia	-27.25	6.50	48.72	12.45	3.91	-18.58	1.52	33.10	5.13	6.45
Daphnia	-31.65	7.19	46.13	11.43	4.03					
Gastropoda	-28.16	2.88	44.45	8.92	4.98	-27.59	4.19	41.54	7.71	5.39
Hemiptera						-16.57	2.35	46.80	11.94	3.92
Hirudinea	-26.51	5.44	46.05	11.86	3.88					
Hirudinea - B						-24.60	5.48	46.99	12.71	3.70
Hyalella						-18.25	-1.27	42.11	7.49	5.62
Hydracarnia						-15.75	0.08	48.51	12.75	3.81
Odonata						-29.84	4.13	47.27	10.76	4.39
Polyphemus pediculus						-18.63	3.73	42.79	7.62	5.62

Appendix 7 continued

			08/07					05/08		
Sample ID	**δ13C**	**δ15N**	**%C**	**%N**	**C/N**	**δ13C**	**δ15N**	**%C**	**%N**	**C/N**
Bosmina	-19.70	0.86	41.22	7.56	5.45					
Calanoida	-19.08	3.47	44.57	8.90	5.01					
Calanoida	-19.39	2.89	48.40	9.39	5.15					
Calanoida	-19.27	2.01	48.15	8.33	5.78					
Chaoborus						-26.80	6.14	51.54	10.32	5.00
Chaoborus						-24.82	6.08	49.87	10.96	4.55
Chaoborus - B						-27.36	6.59	49.34	7.80	6.33
Chironomidae	-17.83	2.51	34.52	6.27	5.51	-28.46	2.96	48.21	10.04	4.80
Chironomidae - B	-24.64	1.27	45.77	12.07	3.79	-33.46	4.49	51.90	9.12	5.69
Cyclopoida	-19.36	2.87	46.41	8.62	5.38	-29.86	3.79	47.21	9.05	5.22
Cyclopoida	-20.12	2.62	47.95	7.50	6.39					
Daphnia	-21.16	3.05	40.08	9.70	4.13	-33.95	2.82	39.18	7.71	5.08
Daphnia	-20.17	2.41	44.42	8.64	5.14					
Ephemeroptera						-26.47	1.41	45.82	11.06	4.14
Gammarus	-20.28	0.84	37.38	7.05	5.30	-27.58	3.32	26.38	4.38	6.03
Gammarus - B						-27.36	1.44	29.41	3.57	8.23
Gastropoda						-31.82	4.87	33.22	5.79	5.73
Hemiptera	-19.56	2.73	50.64	11.59	4.37	-23.25	4.43	47.72	11.70	4.08
Hemiptera -B	-23.78	3.48	48.93	11.44	4.28					
Hirudinea						-26.03	4.92	50.12	9.85	5.09
Hirudinea - B	-19.43	4.97	49.54	14.53	3.41					
Hyalella						-27.00	1.75	27.52	5.80	4.75
Hyallela -B	-23.44	0.52	41.70	9.99	4.18	-28.64	3.27	26.67	4.68	5.71
Hydracarnia										
Hydracarnia - B	-20.77	4.18	47.03	11.71	4.02	-28.02	5.51	48.17	9.04	5.33
Odonata	-19.50	2.77	51.79	11.87	4.36	-28.89	2.94	47.04	7.17	6.56
Odonata	-21.66	3.68	47.17	11.57	4.08	-25.37	5.21	51.53	9.41	5.48
Odonata						-32.45	3.62	45.71	11.16	4.09
Oligochaeta - B	-23.76	1.40	45.16	11.63	3.88	-29.10	5.03	44.68	9.02	4.95
Tricoptera	-16.35	-1.41	41.24	8.11	5.08	-28.92	5.72	35.11	7.39	4.75
Tricoptera - B						-23.78	1.42	51.69	8.85	5.84

Appendix 8

Aquatic invertebrate $\delta^{13}C$, $\delta^{15}N$, %C, %N, and C/N values in PAD 3 during each sampling campaign (month/year). Invertebrates were grouped according to feeding groups and averaged in order to track response to seasonal changes in nutrient availability. An organism with a 'B' at the end of their ID indicates that they were obtained in the benthic samples.

	05/07					07/07				
Organism	**$\delta^{13}C$**	**$\delta^{15}N$**	**%C**	**%N**	**C/N**	**$\delta^{13}C$**	**$\delta^{15}N$**	**%C**	**%N**	**C/N**
Bosmina	-27.64	3.85	43.25	8.32	5.20	-20.82	-0.51	41.41	6.98	5.94
Bosmina						-21.71	0.70	38.85	5.73	6.78
Ceratopongidae	-29.62	2.99	47.20	10.79	4.37	-19.42	-2.46	45.40	9.00	5.05
Chaoborus	-25.96	5.86	45.82	11.38	4.03	-23.34	2.03	42.28	6.66	6.35
Chironomidae	-28.30	2.77	43.37	8.99	4.83					
Chironomidae - B	-33.11	0.71	50.73	9.78	5.19	-27.53	2.71	39.79	9.36	4.25
Cyclopoida	-28.20	3.65	41.38	9.01	4.60	-20.96	1.51	31.16	2.47	12.61
Daphnia	-27.53	3.26	43.25	7.81	5.54	-19.80	0.34	34.26	5.27	6.50
Daphnia	-27.81	3.03	43.23	7.24	5.97	-19.17	0.27	32.22	5.17	6.23
Ephemeroptera						-19.62	-1.90	43.32	7.76	5.58
Ephemeroptera						-19.13	-1.60	43.00	8.63	4.98
Gammarus	-24.06	-0.03	26.41	4.43	5.97	-21.64	-2.82	34.08	3.94	8.65
Gammarus						-20.89	-2.48	31.19	3.50	8.91
Gammarus - B						-24.68	0.48	30.50	4.26	7.16
Gastropoda	-29.97	2.62	44.68	9.66	4.63	-23.99	2.04	44.68	9.66	4.63
Hemiptera						-19.33	2.37	44.48	11.14	3.99
Hirudinea - B	-26.64	4.25	32.93	8.12	4.06	-27.00	4.69	43.11	10.72	4.02
Hyalella						-20.32	-3.10	38.06	4.87	7.81
Hyalella						-21.92	5.67	44.24	11.61	3.81
Hyalella						-20.04	-0.13	30.85	4.86	6.35
Hyalella - B						-26.80	0.94	28.35	4.47	6.34
Hydracarnia						-18.94	2.69	46.37	11.46	4.05
Odonata						-19.54	1.79	48.07	11.96	4.02
Odonata						-19.68	0.70	46.76	9.97	4.69
Odonata						-33.49	1.51	51.87	8.40	6.18
Odonata - B						-28.90	1.16	39.90	7.39	5.40
Oligochaeta - B	-26.39	0.81	50.77	11.85	4.29					
Trichoptera						-21.73	0.11	46.92	11.12	4.22
Trichoptera - B	-21.88	3.77	67.17	18.20	3.69					

Appendix 8 continued

	08/07					05/08				
Organism	**$\delta^{13}C$**	**$\delta^{15}N$**	**%C**	**%N**	**C/N**	**$\delta^{13}C$**	**$\delta^{15}N$**	**%C**	**%N**	**C/N**
Bosmina	-23.65	1.11	40.72	8.14	5.01	-33.50	4.85	40.89	9.04	4.52
Bosmina						-33.58	3.97	44.03	9.48	4.64
Calanoida	-22.23	3.88	49.99	10.14	4.93					
Calanoida	-23.35	3.66	47.91	9.60	4.99					
Chaoborus	-21.22	4.76	51.18	11.14	4.59					
Chaoborus	-21.43	4.94	51.24	11.33	4.52					
Chironomidae	-23.00	1.33	44.37	9.87	4.50	-30.05	3.12	46.67	11.88	3.93
Chironomidae						-38.57	-1.79	44.13	10.76	4.10
Chironomidae - B	-27.45	0.66	45.90	11.95	3.84					
Cyclopoida	-22.14	1.85	40.68	8.62	4.72	-32.51	5.07	39.55	8.14	4.86
Cyclopoida	-22.37	1.99	42.05	9.30	4.52					
Daphnia	-23.06	0.56	42.70	9.15	4.67	-35.45	4.09	45.79	9.33	4.91
Daphnia	-23.37	-0.48	42.07	8.42	5.00					
Ephemeroptera	-20.83	0.71	47.70	11.69	4.08	-31.01	2.47	45.61	11.05	4.13
Ephemeroptera	-22.87	0.24	42.29	10.55	4.01					
Ephemeroptera	-26.02	1.44	40.51	7.76	5.22					
Gammarus	-23.00	-0.47	43.21	7.99	5.40	-29.38	4.65	24.09	5.24	4.60
Gammarus	-23.56	0.87	47.32	9.21	5.14					
Gammarus - B						-27.48	3.07	34.61	7.35	4.71
Gastropoda	-23.27	2.58	45.52	8.88	5.13	-23.27	2.58	44.10	11.04	4.00
Hirudinea	-26.27	4.69	50.07	13.61	3.68	-23.29	3.95	42.12	10.88	3.87
Hyalella	-19.92	0.01	36.08	7.94	4.55	-31.48	2.78	29.59	5.43	5.45
Hyalella	-22.44	-0.27	46.34	10.82	4.28					
Hyalella - B						-27.30	1.01	35.03	9.29	3.77
Hydracarnia - B	-24.41	4.32	51.46	10.37	4.96	-22.87	6.74	43.77	10.68	4.10
Odonata	-22.68	1.82	49.89	9.95	5.01					
Odonata	-21.62	2.20	47.98	10.83	4.43					
Odonata - B	-21.47	2.90	49.08	11.89	4.13	-21.53	4.52	47.52	11.97	3.97
Odonata - B	-25.39	0.53	45.18	9.42	4.79					
Oligochaeta - B	-28.03	1.62	48.70	12.26	3.97	-25.89	1.81	42.14	10.45	4.03
Trichoptera	-19.58	-0.62	49.71	11.35	4.38					
Trichoptera - B	-19.48	-1.56	50.29	10.93	4.60					

Appendix 9

Aquatic invertebrate $\delta^{13}C$, $\delta^{15}N$, %C, %N, and C/N values in PAD 31 during each sampling campaign (month/year). Invertebrates were grouped according to feeding groups and averaged in order to track response to seasonal changes in nutrient availability. An organism with a 'B' at the end of their ID indicates that they were obtained in the benthic samples.

			05/07					07/07		
Organism	**$\delta^{13}C$**	**$\delta^{15}N$**	**%C**	**%N**	**C/N**	**$\delta^{13}C$**	**$\delta^{15}N$**	**%C**	**%N**	**C/N**
Bosmina						-21.48	2.78	36.70	7.57	4.85
Bosmina						-21.56	3.02	36.16	7.62	4.75
Bosmina						-21.46	3.20	37.05	7.70	4.81
Bosmina						-21.07	0.53	40.88	7.65	5.34
Calanoida						-22.52	4.39	45.05	8.56	5.27
Chaoborus						-20.71	-3.14	44.38	5.14	8.63
Chaoborus						-22.25	5.21	46.97	11.81	3.98
Chironomidae	-27.84	2.47	52.25	8.87	5.89	-18.56	0.58	41.18	7.73	5.33
Chironomidae						-18.64	0.39	44.15	8.52	5.18
Chironomidae - B						-28.19	3.70	49.13	9.26	5.31
Cyclopoida	-31.99	7.09	42.40	9.07	4.67	-20.76	2.32	45.28	8.72	5.19
Cyclopoida						-20.39	1.84	44.77	8.46	5.29
Cyclopoida						-19.83	1.39	45.27	8.68	5.21
Daphnia						-21.68	2.85	40.45	8.60	4.71
Daphnia						-21.74	3.16	40.43	8.61	4.69
Daphnia						-21.94	1.76	42.85	7.34	5.84
Daphnia						-21.85	2.52	42.75	8.06	5.30
Ephemeroptera						-25.86	1.35	53.89	9.20	5.86
Gammarus	-24.38	3.80	36.12	8.80	4.10	-22.99	1.43	35.77	5.28	6.78
Gammarus						-23.26	0.15	36.09	4.25	8.49
Gammarus						-21.39	0.63	32.94	4.46	7.39
Gammarus - B						-22.71	1.17	35.26	5.82	6.06
Gastropoda	-21.20	1.59	42.13	12.54	3.36	-31.31	5.71	41.16	8.97	4.59
Hemiptera	-26.02	3.42	47.20	11.93	3.96	-22.82	2.62	47.54	11.78	4.03
Hemiptera						-21.45	2.42	45.30	9.99	4.53
Hirudinea	-21.70	6.59	44.04	13.98	3.15	-23.18	6.51	44.83	11.37	3.94
Hirudinea - B						-25.88	4.73	47.33	12.03	3.94
Hyalella	-28.14	-0.08	27.96	4.78	5.85	-20.84	2.70	37.88	6.85	5.53
Hydracarnia						-18.56	3.50	47.49	12.92	3.67
Odonata	-29.41	4.81	45.14	11.67	3.87	-25.78	2.68	46.61	8.71	5.35
Odonata						-21.15	4.31	45.12	10.76	4.19
Odonata						-19.15	3.39	38.54	10.44	3.69
Odonata - B						-20.15	3.64	48.38	11.57	4.18
Oligochaeta - B	-26.76	2.56	47.74	10.90	4.38	-27.59	3.31	46.59	7.46	6.25
Polyphemus pediculus						-22.36	6.95	45.29	9.01	5.03
Trichoptera	-25.44	2.01	54.99	9.45	5.82	-20.08	0.47	44.74	9.78	4.58

Appendix 9 continued

	08/07					05/08				
Organism	**$\delta^{13}C$**	**$\delta^{15}N$**	**%C**	**%N**	**C/N**	**$\delta^{13}C$**	**$\delta^{15}N$**	**%C**	**%N**	**C/N**
Bosmina	-24.20	0.28	46.09	10.35	4.45					
Bosmina	-24.26	0.61	46.94	10.87	4.32					
Calanoida	-22.21	5.17	47.38	12.77	3.71					
Calanoida	-22.30	5.30	45.02	12.00	3.75					
Chaoborus	-21.05	5.83	46.62	13.17	3.54	-29.04	7.01	53.39	10.04	5.32
Chaoborus	-21.29	5.27	47.18	13.06	3.61	-29.29	7.69	50.12	10.93	4.59
Chaoborus - B	-20.84	4.74	47.08	13.42	3.51	-25.65	6.94	49.42	9.92	4.98
Chironomidae	-24.53	4.13	39.78	10.17	3.91	-28.04	3.59	49.41	11.46	4.31
Chironomidae	-21.32	-0.10	42.70	12.05	3.54					
Chironomidae	-26.11	4.60	33.45	8.45	3.96					
Chironomidae - B	-25.53	3.04	47.17	11.58	4.07	-27.51	3.07	41.96	8.05	5.21
Cyclopoida	-22.98	4.47	44.51	11.60	3.84	-29.30	6.21	44.90	9.20	4.88
Cyclopoida	-22.57	4.44	46.55	12.34	3.77					
Daphnia	-22.86	0.55	44.38	11.15	3.98	-33.34	4.05	40.72	8.41	4.84
Daphnia	-22.63	0.64	45.05	11.42	3.94					
Ephemeroptera						-36.55	3.45	43.04	4.43	9.72
Gammarus	-22.21	-0.04	42.29	9.77	4.33	-27.54	1.72	34.61	7.22	4.80
Gammarus	-22.26	0.19	44.61	10.67	4.18					
Gammarus - B	-23.66	-2.35	42.41	7.16	5.92	-27.09	0.68	39.12	7.34	5.33
Gastropoda						-31.72	3.96	19.86	3.83	5.18
Hemiptera	-25.89	4.85	47.76	11.94	4.00	-26.86	6.84	45.29	11.72	3.86
Hemiptera	-25.53	5.01	48.48	12.06	4.02					
Hirudinea	-23.22	3.09	44.75	12.15	3.68	-23.67	4.66	42.81	8.99	4.76
Hyalella	-23.15	4.52	39.17	9.70	4.04	-28.39	1.95	30.71	6.63	4.63
Hyalella	-25.40	2.16	36.72	7.63	4.81					
Hyalella - B						-27.14	1.22	33.06	6.45	5.13
Hydracarnia	-27.11	4.62	49.55	12.08	4.10					
Hydracarnia - B						-17.23	2.61	46.00	10.91	4.22
Odonata						-22.14	3.30	44.55	10.46	4.26
Odonata						-23.13	5.36	48.12	10.91	4.41
Oligochaeta - B	-26.80	4.83	41.42	11.52	3.60	-27.41	2.43	44.39	9.29	4.78
Plecoptera						-28.49	5.53	43.04	10.66	4.04
Trichoptera						-27.47	2.41	46.81	8.93	5.24
Trichoptera - B	-21.27	-1.23	48.08	11.90	4.04					

Appendix 10

Aquatic invertebrate $\delta^{13}C$, $\delta^{15}N$, %C, %N, and C/N values in PAD 45 during each sampling campaign (month/year). Invertebrates were grouped according to feeding groups and averaged in order to track response to seasonal changes in nutrient availability. An organism with a 'B' at the end of their ID indicates that they were obtained in the benthic samples.

			05/07					07/07		
Organism	**$\delta^{13}C$**	**$\delta^{15}N$**	**%C**	**%N**	**C/N**	**$\delta^{13}C$**	**$\delta^{15}N$**	**%C**	**%N**	**C/N**
Bosmina	-29.12	6.17	31.03	5.94	5.22					
Ceratopongidae	-23.86	3.51	46.90	10.08	4.65					
Chironomidae	-26.15	3.33	45.80	10.48	4.37	-27.01	5.26	41.09	8.93	4.60
Chironomidae - B	-24.42	4.64	42.76	10.15	4.21	-26.77	5.70	41.84	10.77	3.89
Cyclopoida	-30.71	7.22	40.22	9.85	4.08	-31.17	7.22	38.79	8.17	4.75
Cyclopoida	-31.02	6.81	37.80	9.37	4.03					
Daphnia	-31.56	5.43	35.81	7.51	4.77	-30.59	3.05	37.10	7.36	5.04
Daphnia						-30.17	3.74	42.94	8.00	5.37
Ephemeroptera	-26.67	3.94	34.88	6.82	5.11					
Gastropoda						-27.75	4.88	44.68	8.95	4.99
Hemiptera	-22.31	3.57	44.59	12.03	3.71	-28.27	4.69	48.62	12.32	3.95
Hemiptera						-28.27	4.78	49.09	11.97	4.10
Hirudinea - B	-24.90	8.14	47.27	11.30	4.18					
Hyalella	-19.79	3.52	26.24	5.03	5.21	-25.73	1.29	33.31	4.53	7.36
Hyalella						-24.63	1.48	35.94	5.47	6.57
Oligochaeta - B	-26.90	4.86	49.55	10.20	4.86	-25.96	2.35	46.84	12.06	3.88
Trichoptera - B						-27.09	5.80	47.25	11.21	4.22

			08/07					05/08		
Organism	**$\delta^{13}C$**	**$\delta^{15}N$**	**%C**	**%N**	**C/N**	**$\delta^{13}C$**	**$\delta^{15}N$**	**%C**	**%N**	**C/N**
Chironomidae						-33.03	2.16	48.61	10.14	4.80
Chironomidae						-31.18	3.46	48.18	10.71	4.50
Chironomidae - B	-25.30	5.96	41.18	11.14	3.69	-28.62	5.87	47.68	11.01	4.33
Chironomidae - B	-24.53	4.13	47.69	13.07	3.65					
Chironomidae - B	-25.43	5.69	47.69	13.07	3.65					
Daphnia	-28.84	4.53	33.59	8.24	4.08					
Ephemeroptera						-29.73	5.26	41.80	10.77	3.88
Ephemeroptera						-28.54	3.18	40.92	9.58	4.27
Hemiptera						-19.41	2.18	46.72	9.72	4.81
Hemiptera - B						-23.87	4.35	46.21	11.83	3.91
Hirudinea	-25.53	7.90	46.05	13.29	3.47	-28.01	5.79	46.12	9.65	4.78
Hyalella - B						-24.69	4.33	34.19	7.84	4.36
Oligochaeta - B	-24.76	4.71	35.68	10.06	3.55					
Oligochaeta - B	-26.71	4.83	37.07	9.38	3.95					
Trichoptera - B	-26.05	5.19	44.11	10.58	4.17					

Appendix 11

Phytoplankton $\delta^{13}C$, $\delta^{15}N$, %C, %N, and C/N values for each of the sampled basins and the two rivers. The $\delta^{13}C$ and $\delta^{15}N$ values were averaged and used to identify how primary producers respond to seasonal nutrient dynamics within the floodplain basins. These samples were not acidified.

Site ID	Sample Period (mm/yy)	Duplicate	$\delta^{13}C$		$\delta^{15}N$		%C		%N		C/N	
			Result	Repeat	Result	Repeat	Result	Repeat	Result	Repeat	Result	Repeat
PAD 1	05/07	1	-32.53	-32.70	4.89	4.75	23.97	21.34	3.21	2.85	7.46	7.48
		2	-32.88		4.89		16.36		2.01		8.15	
		3	-32.86	-32.88	4.94	4.71	30.26	22.72	3.95	2.93	7.66	7.75
	07/07	1	-19.69	-19.57	3.20	2.91	25.56	25.51	3.04	2.92	8.42	8.72
		2	-21.14		1.61		25.78		3.15		8.18	
		3	-19.88		2.29		10.81		1.23		8.79	
	08/07	1	-20.84		2.15		12.27		1.46		8.41	
		2	-22.60		2.78		15.82		1.82		8.70	
		3	-21.78	-21.79	2.50	3.00	12.08	23.33	1.34	2.62	9.02	8.90
	05/08	1	-34.63		3.91		32.67		4.15		7.88	
		2	-33.46	-33.66	4.19	4.09	21.52	17.40	2.78	2.32	7.73	7.49
PAD 3	05/07	1	-27.44		2.17		18.54		2.35		7.91	
		2	-27.68	-27.74	2.32	2.69	16.66	10.43	2.21	1.43	7.52	7.29
		3	-27.72		2.38		17.89		2.42		7.40	
	07/07	1	-18.03	-17.83	-0.28	-0.24	27.84	15.92	2.58	1.38	10.81	11.55
		2	-17.67	-17.60	0.18	-0.22	12.20	12.81	0.99	1.05	12.27	12.21
		3	-18.01		-0.66		16.27		1.40		11.58	
	08/07	1	-22.81	-22.94	1.15	0.88	24.02	17.75	3.20	2.30	7.50	7.71
		2	-23.37	-23.23	1.21	1.20	21.64	24.20	2.96	3.33	7.30	7.27
		3	-22.62		1.53		32.61		4.23		7.71	
	05/08	1	-33.48		4.27		19.71		3.66		5.39	
		2	-33.28		4.69		20.27		3.92		5.17	

Appendix 11 continued

Site ID	Sample Period (mm/yy)	Duplicate	$\delta^{13}C$		$\delta^{15}N$		%C		%N		C/N	
			Result	Repeat	Result	Repeat	Result	Repeat	Result	Repeat	Result	Repeat
PAD 31	05/07	1	-28.40		2.51		11.64		1.42		8.17	
		2	-30.05	-30.16	3.76	3.81	18.16	16.17	2.42	2.16	7.51	7.49
		3	-30.79		3.44		16.49		2.26		7.29	
	07/07	1	-21.26	-21.46	3.05	3.12	28.16	27.45	3.66	3.61	7.69	7.61
		2	-20.47		1.98		26.14		2.38		10.98	
		3	-21.27		3.38		24.05		3.15		7.64	
	08/07	1	-28.81		-0.72		18.07		3.59		5.03	
		2	-29.09		-0.56		13.16		2.70		4.86	
		3	-29.25	-29.06	-1.00	-0.85	29.45	14.00	6.07	2.84	4.85	4.92
	05/08	1	-30.41		3.10		19.52		3.09		6.32	
		2	-31.78		3.67		24.01		4.40		5.45	
PAD 45	05/07	1	-28.32		4.86		9.84		1.21		8.16	
		2	-27.68		4.69		12.79		1.62		7.89	
		3	-27.64		4.55		10.27		1.26		8.17	
	07/07	1	-28.17	-27.32	2.74	2.58	17.67	24.60	2.11	2.59	8.38	9.49
		2	-28.85		2.61		17.59		2.24		7.85	
		3	-28.20		2.81		16.19		2.11		7.67	
	08/07	1	-28.40		2.69		14.58		2.03		7.19	
		2	-26.68	-26.76	2.25	2.34	10.58	11.84	1.18	1.36	8.94	8.68
		3	-28.05		3.22		13.01		1.67		7.81	
	05/08	1	-26.04	-25.5	0.96	0.63	26.96	22.31	1.74	1.40	15.50	15.96
		2	-24.94		0.34		31.67		2.27		13.94	

Appendix 11 continued

Site ID	Sample Period (mm/yy)	Filter #	$\delta^{13}C$		$\delta^{15}N$		%C		%N		C/N	
			Result	Repeat	Result	Repeat	Result	Repeat	Result	Repeat	Result	Repeat
PAD R11	05/07	1	-24.95		2.00		28.85		1.49		19.33	
	05/07	2	-24.63		1.88		30.56		1.65		18.55	
	05/07	3	-24.34	-24.39	1.08	1.59	23.15	19.71	1.43	1.11	16.24	17.78
	07/07	1	-21.99	-21.99	-22.13	1.51	43.19	40.46	2.91	2.73	14.84	14.83
	07/07	1	-22.04	-22.04	-22.01	1.72	27.08	19.91	1.82	1.28	14.90	15.61
	07/07	2	-21.96	-21.75	1.61		47.64	9.04	3.07		15.53	
	07/07	3	-22.20		1.80		45.43		3.00		15.15	
	07/07	3	-22.12	-22.21	1.40	1.42	15.39	15.76	0.99	1.02	15.51	15.53
	08/07	1	-28.13		2.87		36.78		5.29		6.95	
	08/07	2	-28.54	-28.83	3.10	2.57	38.76	12.82	6.34	2.11	6.12	6.06
	08/07	3	-25.88	-25.76		0.38	5.18	19.42		2.41		8.06
	05/08	1	-21.52		3.23		22.27		1.46		15.30	
	05/08	2	-21.96		1.94		19.42		1.28		15.22	
PAD RMAM	05/07	1	-20.20		1.41		29.28		1.72		17.02	
	05/07	2	-20.54		1.60		17.38		0.99		17.61	
	05/07	3	-19.65		1.64		22.88		1.23		18.58	
	07/07	1	-19.23		2.15		39.55		2.15		18.36	
	07/07	2	-17.77	-17.64	2.32		55.00	12.46	2.76		19.91	
	07/07	3	-17.70	-17.61	2.30	2.30	9.37	8.44	4.00	3.05	2.34	2.77
	08/07	1	-24.42	-24.21	2.22	2.60	12.21	35.92	1.08	3.28	11.28	10.97
	08/07	2	-22.82	-22.57	2.74	2.59	33.93	33.74	2.42	2.43	14.00	13.91
	08/07	3	-22.11		2.63		35.53		2.39		14.84	
	05/08	1	-24.76		1.21		27.49		1.95		14.12	
	05/08	2	-23.89		1.73		28.16		1.81		15.58	

Appendix 12

Isotopic compositions of epiphytes growing on macrophytes (indicated in 'Plant ID' column) within the study basins. Replicate analysis was conducted on the same sample filter. A subsample of the epiphytes similar to each basin was used to observe seasonal trends among the basins. Decomposing plants are indicated with a 'D'.

Site ID	Sample Period (mm/yy)	Plant ID	$\delta^{13}C$		$\delta^{15}N$		%C		%N		C/N	
			Result	Repeat	Result	Repeat	Result	Repeat	Result	Repeat	Result	Repeat
PAD 1	05/07	Myriophyllum - D	-24.05		2.40		11.06		1.45		7.61	
	05/07	Typha - D	-25.86		2.41		25.45		3.06		8.31	
	05/07	Ultriculata vulgaris	-35.81		0.84		21.20		2.39		8.87	
	07/07	Ceratophyllum	-22.28		-1.10		22.55		3.12		7.23	
	07/07	Myriophyllum	-22.06		0.26		16.25		2.06		7.89	
	07/07	P. richardsonii	-16.24		0.31		31.71		3.39		9.36	
	07/07	Typha	-27.82		2.80		16.65		2.00		8.35	
	07/07	Typha - D	-29.31		1.59		25.42		2.81		9.06	
	07/07	Ultriculata vulgaris	-32.19		1.82		15.80		1.84		8.57	
	08/07	Myriophyllum	-18.78	-18.81	0.36	0.29	25.00	29.39	4.64	5.46	5.38	5.38
	08/07	P. zosteriformis	-17.92		-2.24		16.64		2.39		6.95	
	08/07	Typha	-28.26	-27.07	1.38	1.96	26.16	27.03	1.98	2.35	13.23	11.49
	08/07	Ultriculata vulgaris	-32.17		1.39		34.87		4.87		7.16	
	05/08	Ceratophyllum	-30.83		0.02		20.84		3.01		6.92	
	05/08	Myriophyllum - D	-27.89	-26.43	0.11	0.29	13.50	29.94	1.52	3.84	8.86	7.79
	05/08	Typha	-28.41	-29.46	4.42	3.69	13.95	44.98	1.64	4.90	8.48	9.18
	05/08	Typha	-29.28		3.69		22.52		2.56		8.80	
	05/08	Typha - D	-31.39	-31.45	5.38	4.73	22.90	30.40	2.25	3.52	10.16	8.64
	05/08	Ultriculata vulgaris	-33.51	-33.47	5.58	5.44	23.26	28.59	2.40	2.90	9.69	9.84
	05/08	Ultriculata vulgaris	-34.14		5.61		29.04		2.96		9.81	

Appendix 12 continued

Site ID	Sample Period (mm/yy)	Plant ID	$\delta^{13}C$		$\delta^{15}N$		%C		%N		C/N	
			Result	Repeat	Result	Repeat	Result	Repeat	Result	Repeat	Result	Repeat
PAD 3	05/07	Myrophyllum	-26.36		0.99		25.60		2.95		8.67	
	05/07	Sedge - D	-32.06		4.14		40.75		3.87		10.54	
	05/07	Typha	-29.00		3.21		24.01		2.35		10.20	
	05/07	Ultriculata vulgaris	-29.95		3.28		10.64		1.27		8.41	
	07/07	Ceratophyllum	-19.83		0.54		29.99		3.65		8.21	
	07/07	Myriophyllum	-19.17		0.16		40.53		4.57		8.87	
	07/07	P. richardsonii	-18.60		-0.89		23.21		2.08		11.14	
	07/07	Typha	-28.48		1.02		23.35		2.64		8.84	
	07/07	Typha - D	-26.99		0.77		20.83		2.73		7.64	
	08/07	Myriophyllum	-21.52		0.60		24.52		3.24		7.57	
	08/07	P. zosteriformis	-19.77		-1.00		35.62		4.45		8.00	
	08/07	Typha	-28.62		1.07		22.05		2.65		8.34	
	08/07	Typha - D	-27.39		-0.49		17.23		1.78		9.69	
	05/08	Ceratophyllum	-27.40	-27.23	1.81	1.71	25.26	19.93	3.59	2.86	7.03	6.98
	05/08	Myriophyllum	-29.78		0.86		34.77		5.14		6.76	
	05/08	Typha	-26.73		2.82		24.73		3.07		8.05	
	05/08	Typha - D	-30.45		2.64		38.67		4.59		8.42	

Appendix 12 continued

Site ID	Sample Period (mm/yy)	Plant ID	$\delta^{13}C$		$\delta^{15}N$		%C		%N		C/N	
			Result	Repeat	Result	Repeat	Result	Repeat	Result	Repeat	Result	Repeat
PAD 31	05/07	Scirpus - D	-31.02		4.76		12.25		0.98		12.56	
	05/07	Scirpus - D	-25.10		0.64		14.04		1.18		11.86	
	05/07	Tphya - D	-27.06		3.26		11.41		0.91		12.51	
	07/07	P. zosteriformis	-18.10		0.24		35.85		4.16		8.62	
	07/07	Typha	-21.26		1.84		20.30		2.14		9.50	
	08/07	Myriophyllum	-20.54		0.17		28.73		4.53		6.34	
	08/07	P. zosteriformis	-18.52		-0.43		43.19		4.53		9.53	
	05/08	P. richardsonii	-23.21		-1.99		22.58		2.99		7.54	
	05/08	Typha - D	-29.22		3.20		19.32		2.05		9.41	
PAD 45	05/07	Myrophyllum	-23.74		3.49		15.13		1.34		11.32	
	05/07	Scirpus	-27.29		4.49		11.03		1.00		11.03	
	05/07	Typha - D	-26.58	-26.71	5.39	5.15	9.06	52.05	0.90	4.83	10.11	10.78
	07/07	Myriophyllum	-22.62		2.12		27.19		2.12		12.80	
	07/07	Typha	-22.93		2.60		20.68		1.75		11.79	
	07/07	Typha - D	-24.69		3.37		22.81		2.07		11.02	
	07/07	Ultriculata vulgaris	-25.20		2.61		34.93		4.32		8.09	
	08/07	P. gramineus	-21.29		1.44		40.31		5.14		7.84	
	08/07	Sparganium fluctuaris	-22.11		0.76		42.47		5.28		8.05	
	05/08	Scirpus - D	-28.50		3.09		28.07		3.34		8.42	
	05/08	Typha - D	-27.20		2.46		34.91		3.17		11.03	

Appendix 13

Comparison of acidified versus non-acidified $\delta^{13}C$ values for a subset of epiphyte samples collected during various sample periods at the four basins. The non-acidified values indicate the range of isotope values for epiphytes growing on macrophytes within the basins. A sample from each basin was used for to compare carbon isotope values before and after acidification. The acidified values are usually within the range of non-acidified values.

Sample ID	Sample Period (mm/yy)	not acidified			acidified
		$\delta^{13}C$ avg	$\delta^{13}C$ max	$\delta^{13}C$ min	δ13C
PAD 1	07/07	-20.07	-19.57	-21.14	-20.21
PAD 3	07/07	-17.83	-17.60	-18.03	-19.59
PAD 31	05/07	-29.85	-28.40	-30.79	-31.4
PAD 31	07/07	-21.15	-20.47	-21.46	-23.97
PAD 45	05/07	-27.88	-27.64	-28.32	-28.88
PAD 45	07/07	-28.13	-27.32	-28.85	-29.39

Appendix 14

Stable Isotope values and % composition for sediment samples of each basin. During May 2007, samples were taken at PAD 31 along the river bank (*), mid-way between Mamawi Creek and the basin (**), and within the basin (***) to observe if there was a difference in isotopic composition between the locations. There does not appear to be a distinguishing difference between the locations and therefore the sediment value used in Figure 12 as the averaged value of the three sampled locations.

Site ID	Sample Period (mm/yy)	$\delta^{13}C$	$\delta^{15}N$	%C	%N	C/N	
PAD 1	07/07	-23.55	-0.12	16.15	1.46	11.10	
	05/08	-28.42	1.02	25.62	2.25	11.37	
PAD 3	05/08	-26.78	-0.21	21.54	2.13	10.10	
	05/08	-26.72	-0.51	19.81	1.91	10.38	
PAD 31	05/07	-26.12	2.68	0.38	0.02	18.31	*
	05/07	-23.87	2.64	2.07	0.12	17.29	**
	05/07	-25.47	2.41	2.17	0.13	16.10	***
	05/08	-25.46	-0.04	26.73	2.26	11.80	
PAD 45	05/07	-24.51	0.66	14.57	1.35	10.81	
	05/08	-26.72	1.15	33.31	1.43	23.22	